

PROCEEDINGS

TALL TIMBERS FIRE ECOLOGY CONFERENCE

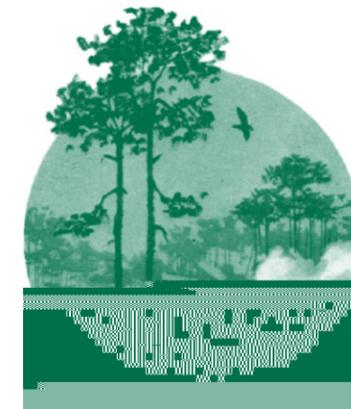
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2007

PROCEEDINGS
23rd

TALL TIMBERS FIRE ECOLOGY CONFERENCE

Fire in Grassland & Shrubland Ecosystems



Edited by
Ronald E. Masters and Krista E.M. Galley

Tall Timbers Research Station
Tallahassee, Florida, USA

ACKNOWLEDGMENTS

A great deal of time, expense, and yes, even blood, sweat, and tears goes into organizing and running a successful conference. A conference is never complete until the ink is dry and the proceedings are finally in hand or on-screen as the case may be. We acknowledge the fantastic support and assistance that made yet another Tall Timbers Fire Ecology Conference a successful endeavor. First and foremost, special thanks go to Kaye Gainey and Lisa Baggett, Administrative Assistants, Tall Timbers Research Station, who worked on all aspects of the conference from initial organization, arrangements, to assistance on-site, organizing the poster session, and sending out papers for review. We also thank Rose Rodriguez, Information Resources Manager, Tall Timbers Research Station, who guided Web site development, published materials, and coordinated arrangements with conference site personnel. We gratefully acknowledge the steering committee members who helped with organizing program topics and session content and the outstanding field trip. They included Bob Hamilton, The Nature Conservancy; Drs. Dave Engle, Terry Bidwell, Sam Fuhlendorf, and John Weir of Oklahoma State University's Rangeland Ecology and Management Program; and Melanie Miller with the USDI Bureau of Land Management. Bob Hamilton deserves special credit and thanks for his organizational efforts on the field trip and for sharing the wonders of the Tallgrass Prairie Preserve with conference participants.

We thank the team that worked behind the scenes with audiovisual and lights: Eric Staller, Tom Ostertag, and Dr. Kevin Robertson, all of Tall Timbers Research Station. We also appreciate our moderators who kept things up-tempo and moving smoothly between presentations and sessions. They included Dr. Terry Bid-

well; Dr. Brian Oswald, Stephen F. Austin University; Dr. Kevin Robertson; Melanie Miller; Dr. Neil Sugihara, USDA Forest Service; and Doug Cram, New Mexico State University.

We give special thanks to our Plenary Speakers, Dr. Winston Trollope, University of Fort Hare, South Africa; Dr. Tom Bragg, University of Nebraska; and Dr. Charles Kay, Utah State University; and to our Banquet Speaker, Dr. Carlton Britton, Texas Tech University. The Seventh E.V. Komarek, Sr. Fire Ecology Lecturer, Dr. Trollope, made a wonderful presentation tying together an important piece of Tall Timbers history with the development of fire ecology in Africa. We appreciate the contributions of our plenary speakers—Dr. Bragg, for his thoughtful talk on contrasts and similarities of grasslands on two continents; and Dr. Kay, for his compelling perspective on the influence of aboriginal fire on North American ecosystems. We also thank Dr. Britton, our banquet speaker, who paid eloquent tribute to one of the great fire ecologists of all time, Dr. Henry Wright.

The 23rd Tall Timbers Fire Ecology Conference Organizing Committee gratefully acknowledges the support of our many sponsors: Tall Timbers Research Station; Joint Fire Science Program; The Nature Conservancy; Rangeland Ecology and Management Program, Oklahoma State University; Southeast Fire Ecology Partnership; USDI Bureau of Land Management; USDI National Park Service; U.S. Fish and Wildlife Service; USDA Forest Service; The Association for Fire Ecology; National Interagency Prescribed Fire Training Center; Oklahoma Chapter, Society for Range Management; and Society for Range Management.

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PREFACE

Ronald E. Masters

Tall Timbers Research Station, 13093 Henry Beadel Drive, Tallahassee, FL 32312, USA

Tall Timbers' Fire Ecology Conferences were begun in 1962 by Tall Timbers Research Station to provide an international forum for discussion of research and research needs in the area of fire ecology and to promote the use of fire as a management tool. Early on, these conferences provided the synergy needed to promote recognition of fire's natural role in the environment. Indeed, the early conferences provided the focal point for an emerging new field—fire ecology. Winston Trollope's paper in these proceedings confirms this. Prior to the convening of these conferences, very few papers were published on fire. Scientists and managers were often ostracized if the topic of fire was broached in a positive light. The times, "they have changed"—a recent Google search on the Internet pulled up 366,000 hits on "fire ecology." Yet the vast majority of the urban public still fails to see the essential role of fire in the environment.

On a periodic basis, conference themes have focused on fire in specific ecosystems where research has brought new knowledge to light and where open scientific discourse would help refine research and management direction and thought. At other times, they have incited paradigm shifts. The proceedings of these conferences are unique in that they are peer-reviewed. This 23rd conference continues the long-standing tradition begun by Ed and Roy Komarek.

The objectives of the 23rd Tall Timbers Fire Ecology Conference were to focus on the ecology of fire in grasslands and shrublands and on grassland-woodland

complexes, which are some of the most rapidly declining systems globally. These systems developed under the interactive influence of fire and grazing and browsing animals—a fact often lost to or ignored by plant ecologists. Large native herbivores are gone from the landscapes of North America; some would argue that domesticated species can fill a surrogate role. The steering committee members have had numerous running discussions over the past decade about how various grazing niches interacted with fire along environmental gradients and at the landscape level. As grassland species of wildlife have declined, it has become increasingly clear that the interactive influence of these forms of natural disturbance is essential to perpetuate a number of habitat-specialist grassland species. Because many of these ideas were being tested on The Nature Conservancy's Tallgrass Prairie Preserve in Northeast Oklahoma, this location came to the forefront as an ideal venue for the 23rd Conference. This conference specifically highlights new research regarding native grazing and patch-burning interactions. Another conference goal was to increase the awareness of current invasive plant problems in these systems and how fire may be used in control efforts. As always, a major objective of these conferences was to facilitate interaction between scientists and land managers to the end of application of research-based knowledge on fire-managed working landscapes. There were 37 oral and 19 poster presentations during the conference, and 166 participants from 27 states and 7 countries.

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SEVENTH E.V. KOMAREK, SR. FIRE ECOLOGY LECTURER: WINSTON S.W. TROLLOPE

Ronald E. Masters

Tall Timbers Research Station, 13093 Henry Beadel Drive, Tallahassee, FL 32312, USA

Winston Trollope is a native South African whose ancestors settled in South Africa from England in 1820. He was born in 1940 and grew up on a farm in the Komga district in the Eastern Cape Province of South Africa. After completing his schooling, he enrolled at the University of Natal in Pietermaritzburg in 1959, where he obtained a BSc. Agric. (1962), MSc. Agric. (1971), and Ph.D. (1984) in rangeland science.

Dr. Trollope's interest in fire ecology was kindled by Professor J.D. Scott at the University of Natal, who was one of the pioneers in fire research in South Africa. Arising from this interest and as part of his responsibilities as Pasture Officer in the Ciskei region of South Africa, he identified the encroachment of undesirable plant species into natural rangeland as one of the main problems facing the livestock industry in this region. This focus formed part of his MSc. project, for which he studied the encroachment and control of macchia vegetation in the mountainous areas of the Eastern Cape Province. He concluded that fire was the most effective, practical, and economic method of controlling macchia vegetation. Subsequently he was able to develop burning programs that continue to be successfully applied throughout the mountain ranges of this region. He was awarded an MSc. (Agric.) in 1971 for this work and received international recognition by being invited to deliver a paper at the Tall Timbers Fire Ecology Conference on "Fire in Africa" held in Tallahassee, Florida, in 1971. Attending this confer-

ence proved to be a turning point in both his research career and fire ecology in South Africa, as it became apparent to him that those in Africa had completely ignored the effects of type and intensity of fire on the vegetation, having focused only on the effects of season and frequency of burning. This realization led to an investigation, as part of his Ph.D., on the effects of type and intensity of fire on the grass and tree components of the vegetation in African savannas. This research has resulted in a greater understanding of the role of fire in savanna ecosystems and a significant improvement in the use of fire as a range management practice for both domestic livestock systems and wildlife management.

Since the recent political changes in South Africa, Dr. Trollope has been fortunate enough to be able to extend his research interests farther afield in Africa. He has initiated studies on the fire ecology of grassland and savanna ecosystems in the central highlands of Kenya, the Ngorongoro Crater and Serengeti in Tanzania, the Caprivi region of Namibia, the Gile National Reserve in Mozambique, and the Okavango Delta in Botswana. One of the highlights has been traveling to the Konza Prairie Biological Station in Kansas, USA, in 2003, where with his wife, Lynne, and two students from the University of Fort Hare, he conducted a project comparing the behavior of fires in tallgrass prairies with the fire behavior in African grasslands and savannas.

SEVENTH E.V. KOMAREK, SR. MEMORIAL FIRE ECOLOGY LECTURE

FIRE—A KEY FACTOR IN THE ECOLOGY AND MANAGEMENT OF AFRICAN GRASSLANDS AND SAVANNAS

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Seventh E.V. Komarek, Sr. Memorial Fire Ecology Lecturer

ABSTRACT

Fire is regarded as a natural ecological factor of the environment that has been occurring since time immemorial in savannas and grasslands on the continent of Africa. The African continent is highly prone to lightning storms and has an ideal fire climate with dry and wet periods. It also has the most extensive area of tropical savanna on the earth, characterized by a grassy understory that becomes extremely flammable during the dry season. The use of fire in the management of vegetation for both domestic livestock systems and wildlife is widely recognized. Research on the effects of fire has been conducted throughout the grasslands and savannas of Africa since the early 20th century, and has focused on the effects of season and frequency of burning on forage production potential of the grass sward and the ratio of bush to grass in African savannas. In 1971, a conference was held in the United States by Tall Timbers Research Station in Tallahassee, Florida, on the theme “Fire in Africa.” This congress was attended by fire ecologists from throughout Africa. A major benefit that accrued from this meeting was the realization that in Africa the study of fire behavior and its effects on the ecosystem, as described by type and intensity of fire, had been largely ignored in the fire research. This led to recognition that the effects of fire must include the effects of all the components of the fire regime on the ecosystem, viz., the type and intensity of fire and the season and frequency of burning. As a consequence, a research program was initiated in South Africa in 1972, and later extended to East Africa in 1992, to characterize the behavior of fires burning in savanna and grassland vegetation and to determine the effect of type and intensity of fire on the vegetation. This research program has successfully developed a greater understanding of the effects of type and intensity of fire in African grasslands and savannas. This in turn led to the development of more effective and practical guidelines for fire regimes to be used in controlled burning for domestic livestock and wildlife management systems in grasslands and savannas. Research on the fire ecology of grasslands and savannas continues to draw scientific attention in Africa, and currently there are active research programs being conducted in the Eastern Cape Province, KwaZulu-Natal Province, and Kruger National Park in South Africa, and in the Ngorongoro Crater in Tanzania.

keywords: Africa, fire, fire behavior, fire effects, grassland, prescribed burning, savanna.

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INTRODUCTION

Fire is regarded as a natural ecological factor of the environment in Africa that has been occurring since time immemorial. It is estimated that human beings have used fire for more than 1 million y, and in Africa has extended grasslands and savannas at the expense of evergreen forests (Bond and van Wilgen 1996). The continent of Africa is highly prone to lightning storms and has an ideal fire climate comprising dry and wet periods. It also has the most extensive area of tropical savanna on the earth, characterized by a grassy understory that becomes extremely flammable during the dry season (Komarek 1971). The use of fire in the management of vegetation for both domestic livestock systems and wildlife management is widely recognized. Research on the effects of fire has been conducted throughout the grassland and savanna areas of Africa since the early 20th century (West 1965) and focused on the effects of season and frequency of burning on forage production potential of the grass sward and the ratio of bush to grass in African savannas. However, in 1971 a conference was convened by Tall Timbers Research Station in Tallahassee, Florida, with the theme "Fire in Africa." This congress was attended by fire ecologists from throughout Africa. A major benefit that accrued from this meeting was the realization that in Africa the study of fire behavior and its effects on the ecosystem, as described by type and intensity of fire, had been largely ignored up until that time. This was a turning point in research on the fire ecology of African grasslands and savannas and led to the further recognition that the effects of fire must include the effects of all the components of the fire regime on the ecosystem, viz., the type and intensity of fire and the season and frequency of burning. As a result, a research program was initiated in South Africa in 1972 (Trollope 1978, Trollope and Potgieter 1985), and later extended to East Africa in 1992 (Trollope and Trollope 1999), to characterize fire behavior in savanna and grassland vegetation and to determine the effect of type and intensity of fire on vegetation. This research program has successfully developed a greater understanding of the influence of fire type and intensity in African grasslands and savannas (Trollope 1984, Trollope and Tainton 1986, Trollope and Trollope 1999, de Ronde et al. 2004). This led to the development of more effective and practical guidelines for fire regimes to be used in controlled burning for domestic livestock and wildlife management systems in grassland and savanna areas. Research on the fire ecology of grasslands and savannas continues to enjoy scientific attention in Africa, and currently the author is associated with active research programs in Eastern Cape Province, KwaZulu-Natal Province, and Kruger National Park in South Africa, and in the Ngorongoro Crater in Tanzania.

This paper presents an overview of the beneficial and significant effects that the 11th Tall Timbers Fire Ecology Conference (Tallahassee, Florida; 22–23 April 1971) has had on a greater understanding of the effects and use of fire in African grasslands and sa-

vannas. It is also a tribute to the vision and foresight that E.V. Komarek, Sr. had into the ecology of fire-dependent ecosystems and his willingness and passion to share his knowledge with society in general and in particular with communities reliant on the ecological well-being of African grasslands and savannas.

FIRE BEHAVIOR

The effect of fire on natural ecosystems involves the response of living organisms to the release of heat energy through the combustion of plant material. The manner in which and the factors that influence release of heat energy involve the study of fire behavior. Fire behavior is defined as the release of heat energy during combustion as described by fire intensity, rate of spread of the fire front, flame characteristics, and other related phenomena (Trollope 1981). In Africa there is a serious deficiency of knowledge concerning fire behavior, and limited efforts have been made to quantify the factors that influence fire behavior. Also, no attention has been paid to the dynamics of heat energy release and its subsequent effect on the ecosystem. Basically, the effect of fire on plants depends upon the amount of heat energy released, rate of release, and vertical level at which heat energy is released. The determination of such relationships helps clarify many of the apparently inexplicable effects of fire often cited in the literature. The following fire behavior parameters have been developed and identified that quantitatively describe fire behavior and its effects on savanna and grassland vegetation.

Available Heat Energy

The total amount of heat energy contained per unit mass of fuel is called the *heat of combustion*, but not all this total potential heat energy is released during a fire because some heat energy remains in unburned plant material. The net heat energy released during a fire is called the *heat yield*. Heat yield was determined for fine grass fuels in savannas in the Eastern Cape Province, South Africa, and values of 16,890 and 17,781 kJ/kg were obtained for grass fuels burning as head- and backfires, respectively (Trollope 1983). These heat yields are similar to heat yields quoted and used in the United States and Australia (Albini 1976, Luke and McArthur 1978) and were subsequently used for estimating available heat energy in all fire behavior studies conducted in southern and East Africa. In practice, the available heat energy is estimated as the plant fuel load available for combustion during a fire and is expressed in kilograms per hectare or per square meter.

Rate of Heat Energy Release

Fire intensity refers to the rate at which heat energy is released during a fire and is defined as the release of heat energy per unit time per unit length of fire front ($\text{kJ s}^{-1} \text{m}^{-1}$; Byram 1959). Numerically, fire intensity is the product of the available heat energy, the rate of spread of the fire front, and the mass of fuel consumed, which can be expressed as the equation

Table 1. The range of conditions under which controlled burns were applied in the Eastern Cape Province and Kruger National Park, South Africa (Trollope 1983, Trollope and Potgieter 1985). Copyright © 2002 Millpress Science Publishers. From *Forest Fire Research & Wildland Fire Safety*, by W.S.W. Trollope, L.A. Trollope, and D.C. Hartnett, "Fire behaviour a key factor in African grasslands and savannas." Reprinted by permission of Millpress Science Publishers and the authors.

Variable	Mean	Min.	Max.
Fuel load (kg/ha)	3,847	1,152	10,500
Fuel moisture (%)	32.1	7.5	68.8
Air temperature (°C)	23.8	14.3	35.8
Relative humidity (%)	36.6	4.2	82
Wind speed (m/s)	2.6	0.3	6.7
Fire intensity (kJ s ⁻¹ m ⁻¹)	2,566	136	12,912

$$I = Hwr,$$

where I = fire intensity (kJ s⁻¹ m⁻¹), H = heat yield (kJ/kg), w = mass of fuel consumed (kg/m²), and r = rate of spread of the fire front (m/s).

Albini (1976) stated that Byram's description and definition of fire intensity has proved to be very useful in fire behavior studies and quotes van Wagner (1973), who found that fire intensity was significantly correlated with the height of lethal scorching of coniferous tree crowns. Research in the Eastern Cape Province and Kruger National Park, South Africa, supports this view, as fire intensity had a highly significant positive effect on the top-kill of stems and branches of trees and shrubs in these two areas (Trollope and Tainton 1986, Trollope 1999).

Vertical Distribution of Heat Energy

A reliable indicator of the vertical distribution of heat energy released during a fire is the perpendicular height of the flames from ground level, i.e., flame height. Research in the Eastern Cape Province has shown that flame height has a significantly positive effect on the top-kill of stems and branches of trees at increasing height levels (Trollope 1983).

Factors Influencing Fire Behavior

Subsequent research showed that fire intensity and flame height were significantly correlated and both had significant effects on vegetation. Research also showed that fire intensity can best be used for describing the general behavior of fires and their effects on vegetation; consequently, this parameter was focused on in modeling the behavior of fires and their effects on vegetation. Factors influencing fire behavior were investigated in terms of those variables that need to be considered when applying controlled burns as a range management practice: fuel load, fuel moisture, relative humidity, air temperature, wind speed, and slope (Brown and Davis 1973, Luke and McArthur 1978, Cheney 1981, Leigh and Noble 1981, Shea et al. 1981, Wright and Bailey 1982). The effect of these variables on fire intensity were investigated in the Eastern Cape Province and Kruger National Park (Trollope 1978, Trollope 1983, Trollope and Potgieter 1985), where

Table 2. Effects of fuel load, fuel moisture, relative humidity, and wind speed on fire intensity ($n = 200$).

Independent variable	Transformation	Coefficient of determination (R^2)	Effect (%)	Significance
Fuel load	x	0.427	42.7	$P \leq 0.01$
Fuel moisture	\sqrt{x}	0.574	14.7	$P \leq 0.01$
Relative humidity	x^2	0.584	1.0	$P \leq 0.02$
Wind speed	$1/x$	0.600	1.6	$P \leq 0.02$
Total			60.0	

235 controlled burns were applied to areas ranging from 0.2 to 7 ha. The range of conditions under which the controlled burns were applied is presented in Table 1.

The influence of these environmental variables on fire behavior was determined by multiple regression analysis of data recorded during controlled burns where fuel load, fuel moisture, relative humidity, and wind speed were independent variables and were regressed against fire intensity as the dependent variable. Air temperature was not included in the analysis because of correlation with relative humidity, thereby not fulfilling the criteria of independence. The effect of slope was not considered because all burns were applied to relatively flat terrain.

All the independent variables had a statistically significant effect and accounted for 60% of the variation in fire intensity (Table 2). Fuel load and fuel moisture influenced fire intensity to the greatest extent, whereas relative humidity and wind speed had a significant but far smaller effect. The resulting multiple regression equation for predicting fire intensity is

$$FI = 2,729 + 0.8684x_1 - 530\sqrt{x_2} - 0.1907x_3^2 - 596(1/x_4),$$

where FI = fire intensity (kJ s⁻¹ m⁻¹), x_1 = fuel load (kg/ha), x_2 = fuel moisture (%), x_3 = relative humidity (%), and x_4 = wind speed (m/s). The regression equation is based on the following statistics: number of cases = 200, multiple correlation coefficient (R) = 0.775 ($P \leq 0.01$), and coefficient of determination (R^2) = 0.600.

The model was tested with independent fire behavior data and found to be highly significant ($r = 0.749$, $df = 33$, $P \leq 0.01$) and accounted for 56% of the variation in fire intensity. By normal statistical standards, this coefficient of determination is rather low, and generally regression equations are used for predictive purposes only when the coefficient of determination accounts for at least 95% of the variation in the dependent variable. However, experience gained during this research suggests that it is virtually impossible to attain these levels of precision when modeling such a complex and inherently variable phenomenon as fire behavior; subsequent experience with this model showed that it provided adequate precision for formulating practical guidelines for controlled burning in grasslands and savannas both in southern and East Africa.

FIRE EFFECTS

Fire ecology refers to the response of the biotic and abiotic components of an ecosystem to a fire regime. This would include type and intensity of fire and season and frequency of burning (Trollope et al. 1990). Following the 11th Tall Timbers Fire Ecology Congress in 1971, a research program was initiated in South Africa in 1972 to determine the effect of all the components of the fire regime on vegetation, i.e., effects of type and intensity of fire and season and frequency of burn. Unfortunately, a similar research program was not known to be initiated elsewhere in Africa until 1992. An overview follows of known effects of the fire regime on grass and bush vegetation in African grasslands and savannas, based largely on the results of this research program.

Type of Fire

The most common types of fire in grassland and savanna areas are surface fires (Trollope 1983) burning either as head- or backfires. Crown fires do occur in savannas, but only under extreme conditions. Generally under these conditions they occur as passive crown fires characterized by the “torching” of individual trees rather than as active crown fires that are sustained by more abundant and continuous aerial fuels. Thus, type of fire determines the vertical level at which heat energy is released in relation to the location of meristematic tissues.

Trollope (1978) investigated the effects of surface fires, occurring as either head- or backfires, on the grass sward in the arid savannas of the Eastern Cape. The results showed that backfires significantly ($P \leq 0.01$) depressed regrowth of grass in comparison to headfires because a critical threshold temperature of approximately 95°C was maintained for 20 s longer during backfires than during headfires. Also, more heat was released at ground level during backfires compared to during headfires; therefore, shoot apices of the grasses were more adversely affected by backfires.

Bush (i.e., woody, shrubby vegetation) is very sensitive to various types of fires because of differences in the vertical distribution of heat energy release. Field observations in Kruger National Park and in Eastern Cape Province indicate that crown and surface headfires cause the highest amount of top-kill of stems and branches compared to that of backfires. Unfortunately, there are only limited quantitative data to support these observations. Research results were obtained from a burning trial at the University of Fort Hare in the False Thornveld of the Eastern Cape (arid savanna), where a field-scale burn was applied to a 62-ha area to control bush encroachment. The majority of the trial area was ignited as a headfire, and results showed that the phytomass of bush was reduced by 75% in the area burned as a headfire compared to 42% in the area burned as a backfire. This happened because the flame height of headfires can be up to 3 times greater than that of backfires, resulting in higher temperatures being generated above ground level (Trollope 1978). Therefore, the aboveground growing points of these

plants, located in the canopies of trees and shrubs, are subjected to greater heat loads and resultant damage during headfires than during backfires. Similar results were obtained in the Scattered Tree Grassland (*Acacia-Themedea* [Edwards and Bogdan 1951]) range type in the Kenya central highlands where the effects of head- and backfires on the top-kill of savanna trees and shrubs bush were investigated (see figure 3 in Trollope et al. 2002).

Head- and backfires had significantly different effects on top-kill of trees and shrubs, with headfires generally causing a greater top-kill than backfires (see figure 3 in Trollope et al. 2002). Initially both types of fires resulted in a high top-kill of stems and branches when the bush was short, but as trees and shrubs increased in height, backfires caused a lower top-kill compared to that of headfires. This trend became more pronounced with trees >2 m in height. Headfires generate greater flame heights than backfires; therefore, fire-susceptible growing points of taller trees and shrubs are above the flaming zone of combustion during backfires compared to during headfires.

Fire Intensity

The effect of fire intensity on the recovery of the grass sward after burning was investigated in arid savannas of the Eastern Cape Province. After a series of fires ranging in intensity from 925 to 3,326 kJ s⁻¹ m⁻¹ (cool to extremely intense fires), no differences were detected in the recovery of the grass sward at the end of the first or second growing seasons after the burns (Trollope and Tainton 1986). Therefore, fire intensity has little influence on the recovery of the grass sward after a burn. This is a logical result because otherwise intense fires would not favor development and maintenance of grasslands.

The effect of fire intensity on trees and shrubs has been studied in arid savannas in Eastern Cape Province (Trollope and Tainton 1986) and Kruger National Park (Trollope et al. 1990). Plant mortality and total top-kill of stems and branches of bush at different heights was determined. Bush is very resistant to fire alone; in the Eastern Cape Province, the mortality of trees and shrubs after a high-intensity fire of 3,875 kJ s⁻¹ m⁻¹ was only 9.3%. In Kruger National Park, the average mortality of 14 of the most common bush species subjected to fires ranging in intensity from 110 to 6,704 kJ s⁻¹ m⁻¹ ($n = 43$) was only 1.3%. In both areas, the majority of the trees that suffered a top-kill of stems and branches coppiced from the collar region of the stem (Figure 1). A significantly greater top-kill of bush with increasing fire intensities was noted. However, the bush became more resistant to fire as the height of the trees and shrubs increased (Figure 2). Similar responses were reported in arid savannas of Eastern Cape Province (Trollope and Tainton 1986) and in the Scattered Tree Grassland (*Acacia-Themedea* savanna) in the Kenya central highlands (Trollope and Trollope 1999).

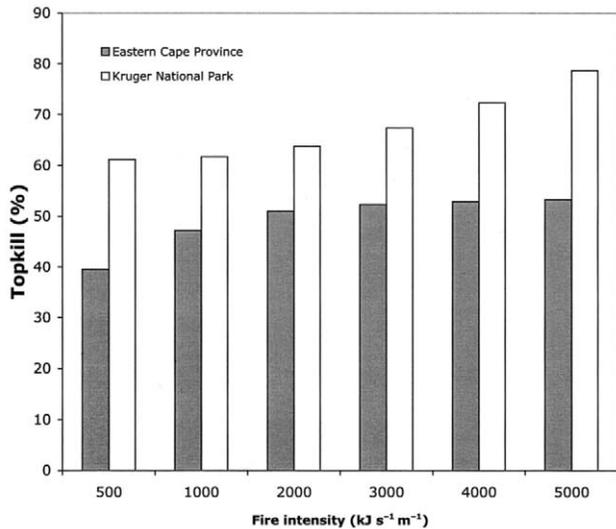


Fig. 1. Effect of fire intensity on the top-kill of trees and shrubs 2 m high in the Eastern Cape Province and Kruger National Park, South Africa. Copyright © 2002 Millpress Science Publishers. From *Forest Fire Research & Wildland Fire Safety*, by W.S.W. Trollope, L.A. Trollope, and D.C. Hartnett, "Fire behaviour a key factor in African grasslands and savannas." Reprinted by permission of Millpress Science Publishers and the authors.

Season of Burning

Very little published quantitative information is available on the effect of season of burning on the grass sward. West (1965) stressed the importance of burning when grass is dormant. Scott (1971) quoted data from the Southern Tall Grassveld of KwaZulu-Natal Province, South Africa, where mean grass basal cover of plots burned in autumn, late winter, and after the first spring rains for a period exceeding 20 y, was 12.8, 13.0, and 14.4%, respectively. The absence of large differences in mean basal cover obtained with these different seasons of burning indicated that, for all practical purposes, burning when the grass sward is dormant has very little influence on the grass sward. This conclusion is supported by Tainton et al. (1977), Dillon (1980), and Everson et al. (1988), who also found that burning before or immediately after the first spring rains in KwaZulu-Natal Province had essentially the same effect on the recovery of a burned grass sward. Conversely, with grasslands burned later in the season when actively growing, causes a high mortality of tillers of *Themeda triandra*, resulting in a significant reduction in abundance (Dillon 1980, Everson et al. 1988).

The effect of season of burning on the recovery

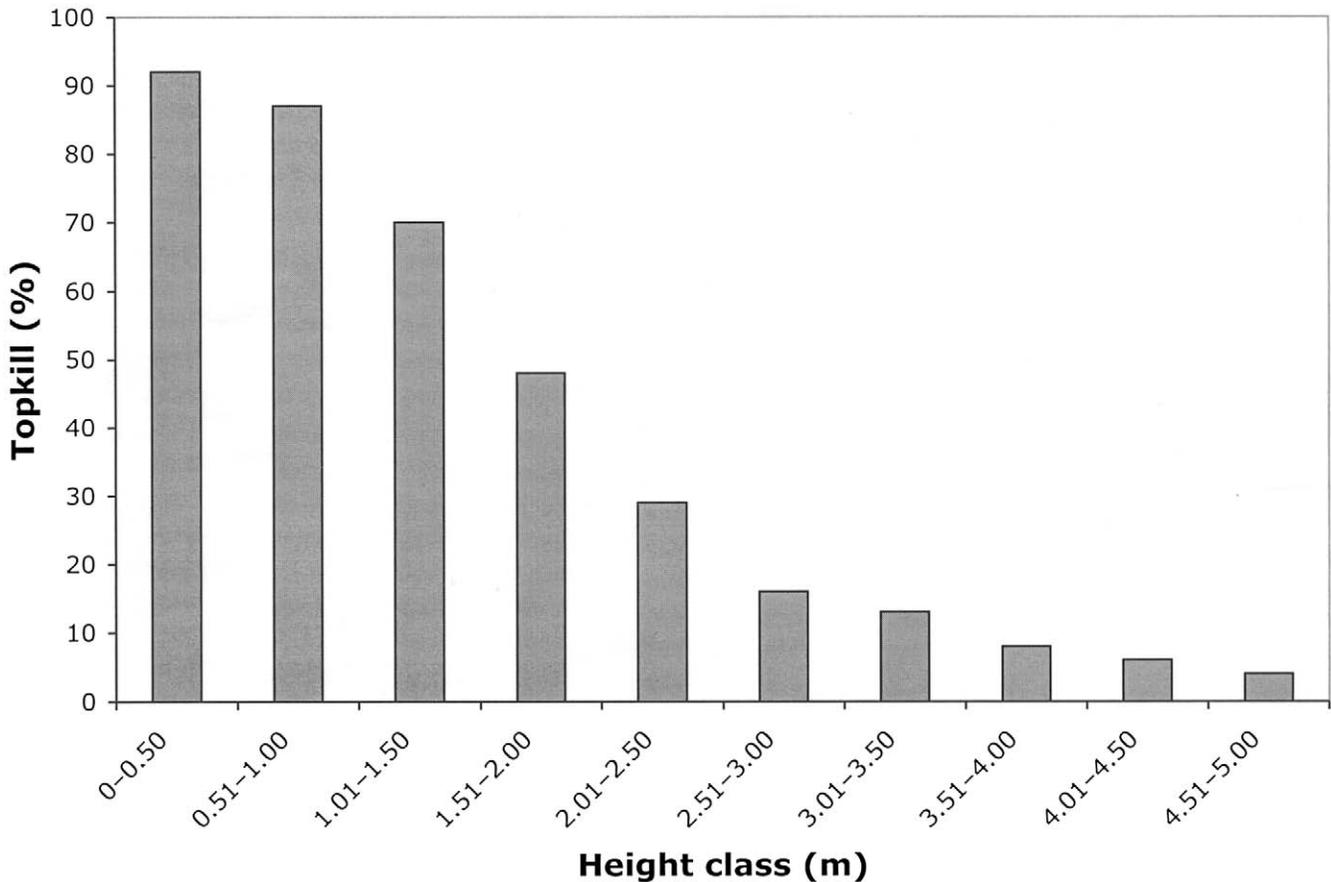


Fig. 2. Effect of height of trees and shrubs on the top-kill of bush subjected to a fire intensity of $3,000 \text{ kJ s}^{-1} \text{m}^{-1}$ in Kruger National Park, South Africa. Copyright © 2002 Millpress Science Publishers. From *Forest Fire Research & Wildland Fire Safety*, by W.S.W. Trollope, L.A. Trollope, and D.C. Hartnett, "Fire behaviour a key factor in African grasslands and savannas." Reprinted by permission of Millpress Science Publishers and the authors.

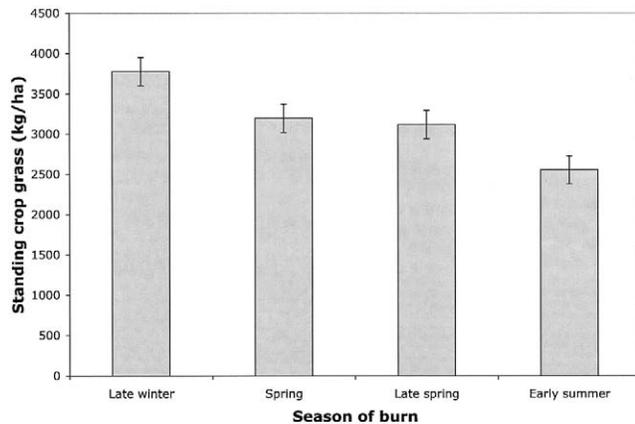


Fig. 3. Effect of season of burning on the standing crop of grass sward (kg/ha) in arid savannas of the Eastern Cape Province, South Africa (Trollope 1987). Copyright © 1987 Grassland Society of Southern Africa. From *Journal of the Grassland Society of Southern Africa*, by W.S. Trollope, "Effect of season of burning on grass recovery in the false thornveld of the eastern Cape." Reprinted by permission of National Inquiry Services Centre.

of grass was also investigated in the arid savannas of the Eastern Cape Province (Trollope 1987). To determine the effect of burning the grass sward in late winter, spring, late spring, and early summer, the standing crop during the first post-fire growing season was measured.

Burning in late winter consistently resulted in a significantly greater recovery of the grass sward during the first post-fire growing season than the other treatments (Figure 3). Conversely, early summer burns that were applied when the grass was actively growing had a significantly depressive effect ($P < 0.01$) throughout the recovery period on the regrowth of the grass sward in relation to the other treatments. Burning when the grass was actively growing adversely affected recovery of the grass sward when compared with burning when the grass was dormant.

Observations and measurements made during the early summer burn showed that grass tillers were actively growing, and shoot apices were therefore probably elevated and vulnerable to fire. Furthermore, the mean rate of spread for the early summer burns was 0.11 m/s compared to 0.31 m/s for the later winter burns. This would suggest that the slow-moving early summer burn resulted in a longer duration of critical threshold temperatures compared to the fast-moving winter burn and therefore caused more damage to exposed shoot apices (Trollope 1987). Subsequent investigations have confirmed this and found that the rate of spread was significantly positively correlated ($r = 0.2669$, $df = 54$, $P \leq 0.02$) with recovery of the grass sward during the first post-fire growing season. These different sources of evidence lend support to the view that the effect of season of burning on the grass sward depends upon the physiological state of the grass at the time of the fire.

Season of burning also has an effect on the botanical composition of the grass sward. In KwaZulu-Natal Province, *Themeda triandra* declined after burning in

autumn compared to burning in winter and spring, whereas *Tristachya leucothrix* increased (Bond and van Wilgen 1996).

The effect of season of burning on savanna trees and shrubs generally is confounded by fire intensity. When the trees are dormant in winter the grass is dry and supports intense fires, whereas when the trees are actively growing during summer the grass is green and fires are much cooler. West (1965) postulated that trees and shrubs are probably more susceptible to fire at the end of the dry season when plant reserves are depleted because of new spring growth. However, Trollope et al. (1990) showed that mortality of bush in Kruger National Park was only 1.3% after fires that had been applied to bush ranging from dormant to actively growing plants. Therefore, it would appear that bush is not sensitive to season of burn.

Frequency of Burning

The effect of frequency of burning on vegetation is influenced by event-dependent effects and interval-dependent effects (Bond and van Wilgen 1996). Event-dependent effects occur at the time of the fire and are influenced by type and intensity of the burn and the physiological state of the vegetation at the time of the fire. Interval-dependent effects are influenced by treatment and growing conditions that occur during the interval between burns. These two overall effects tend to confound interpretation of the effect of frequency of burning and must be considered when reporting on the effect of frequency of burning.

Frequency of burning has a marked effect on botanical composition, with species such as *Themeda triandra* being favored by frequent burning and *Tristachya leucothrix* being favored by infrequent burning in moist grasslands of KwaZulu-Natal Province (Scott 1971, Dillon 1980). Similar results have been obtained in arid savannas of Eastern Cape Province, where frequent burning favored an increase in *Themeda triandra* and a decrease in *Cymbopogon plurinodis* (Robinson et al. 1979). In East Africa, Pratt and Gwynne (1977) reported that *Themeda triandra* is a common constituent of grasslands in the Kenya central highlands on undulating plateaus and mountain flanks where fires are regular occurrences and grazing pressure is not too high. Where fires are infrequent or lacking, upland grassland tends to become dominated by *Pennisetum schimperii* and *Eleusine jaegeri*, which are coarse-tufted species of little forage value. These are interval-dependent effects of frequency of burning because *Themeda triandra* is sensitive to low light conditions that develop when the grass sward is not defoliated, and this species rapidly becomes moribund during extended intervals between fires. Conversely, species such as *Tristachya leucothrix* and *Cymbopogon plurinodis* are not as sensitive to low light conditions and survive extended periods of non-defoliation.

Conflicting results have been obtained on the effect of frequency of burning on bush. Kennan (1971) in Zimbabwe and van Wyk (1971) in Kruger National

Park both found that there were no biologically meaningful changes in bush density in response to different burning frequencies. In the False Thornveld of Eastern Cape Province, Trollope (1983) found that after 10 y of annual burning the density of bush increased by 41%, the majority in the form of short coppicing plants. Conversely, Sweet (1982) in Botswana and Boulton and Rodel (1981) in Zimbabwe found that annual burning resulted in a significantly greater reduction in the density of bush than less frequent burning. It is difficult to draw any general conclusions from these contradictory results except to note that in all cases significant numbers of trees and shrubs were present even in the annually burned areas, irrespective of whether they had decreased or increased after burning. These variable results suggest that the effect of frequency of burning on woody vegetation is more an event-dependent effect in which factors like type and intensity of fire have had highly significant individual effects overshadowing the effect of frequency of burning per se.

The withdrawal of fire for extended periods of time appears to have a more predictable effect. For example, on the Accra Plains in southeastern Ghana, protection of moist savanna from fire for 29 y resulted in the development of forest-type vegetation with a fairly closed canopy. The fire-sensitive tree *Ceiba pentandra* became dominant (Carson and Abbiw 1990). Similar results have been obtained in the Lamto Reserve in the Ivory Coast, which receives a high mean annual rainfall of 1,300 mm and forms part of the Guinea savanna immediately adjacent to deciduous rain forest. The savanna vegetation is subjected to annual burning during the middle of the dry season. In a study investigating the exclusion of fire for 13 y, it was found that after 8 y open savanna rapidly changes into a dense, closed formation and after 13 y the first signs of forest developing occurred in the form of seedlings and saplings. In all the burned savannas of the Lamto Reserve, the pressure of forest elements on savanna vegetation is very high and fire exclusion initiates the development of forest (Menaut 1977). Similar trends have been found in the more arid savannas (500–700 mm annual rainfall) in southern Africa where in Kruger National Park fire exclusion caused both an increase in density and size of tree and shrub species (van Wyk 1971).

The effect of frequency of burning on forage production has not been intensively studied in South Africa and only limited quantitative data are available. The general conclusion is that the immediate effect of burning on the grass sward is a significant reduction of grass yield during the first post-fire growing season, but the depressive effect disappears during the second post-fire season (Tainton and Mentis 1984, Trollope 1984).

Frequent fires generally improve and maintain the nutritional quality of grassland, particularly in high rainfall areas, making it highly attractive to grazing animals. This phenomenon has been recorded throughout the savanna and grassland areas of Africa (West 1965, Tainton et al. 1977, Moe et al. 1990, Munthali

and Banda 1992, Schackleton 1992). West (1965) stated that the fresh green shoots of new growth on burned grassland are very high in protein and quotes Plowes (1957) who found that the average crude protein content of 20 grasses after burning at the Matopos Research Station in Zimbabwe was 19%. This is approximately 2 times the protein content of mature grasses that have not been burned at the end of the dry season.

Apparently no information is available on the effect of frequency of burning on production and quality of browse by trees and shrubs in savanna areas.

Interactions between Fire and Herbivory

Post-burn herbivory can have a highly significant effect on botanical composition and structure of vegetation. The arid savannas of Eastern Cape Province receive a mean annual rainfall of approximately 500 mm and are an important livestock production area, particularly cattle ranching. However, during the last 50 y the grazing capacity of the rangeland has been drastically reduced by the encroachment of bush, primarily *Acacia karroo*.

An experiment was initiated in 1972 in the False Thornveld of the Eastern Cape at the University of Fort Hare to determine the role fire can play in controlling bush encroachment. Fire was chosen as a possible solution because ecologically it is recognized as a natural factor of the environment in African savannas and economically it is an indirect cost technique, making it suitable for the rehabilitation of rangeland with an inherently low economic potential. The hypothesis that fire maintains trees and shrubs at an available height and in an acceptable state for browsing animals was tested. The treatments involved applying a high-intensity fire ($3,875 \text{ kJ s}^{-1} \text{ m}^{-1}$) to a moderately encroached area of savanna (2.2 ha) dominated by *Acacia karroo* with a dense grass sward dominated by *Themeda triandra*. The high-intensity fire caused an 81.5% top-kill of tree stems and branches, but only 9.9% were killed (Trollope 1974). After the fire, follow-up treatments were superimposed on the burned area and have been maintained to date (33 y):

- 1) Grazing treatment (0.6 ha)—heavy grazing with cattle after the first frost in early winter (May–June) until uniform utilization of the grass sward has been achieved (approximately 10 d);
- 2) Browsing–grazing treatment (1 ha)—continuous stocking with goats (1 small stock unit/ha) and heavy grazing with cattle after the first frost in early winter (May–June) until uniform utilization of the grass sward has been achieved (approximately 10 d);
- 3) Burning treatment (0.6 ha)—annual burning at the end of winter prior to the first spring rains (July–August) (Trollope and Dondofema 2003).

These follow-up treatments have been maintained since the initiation of the experiment in 1972, and the long-term effects of the grazing, browsing–grazing, and annual burning treatments on the density of the bush were assessed in 2001 (Figure 4). Initially in

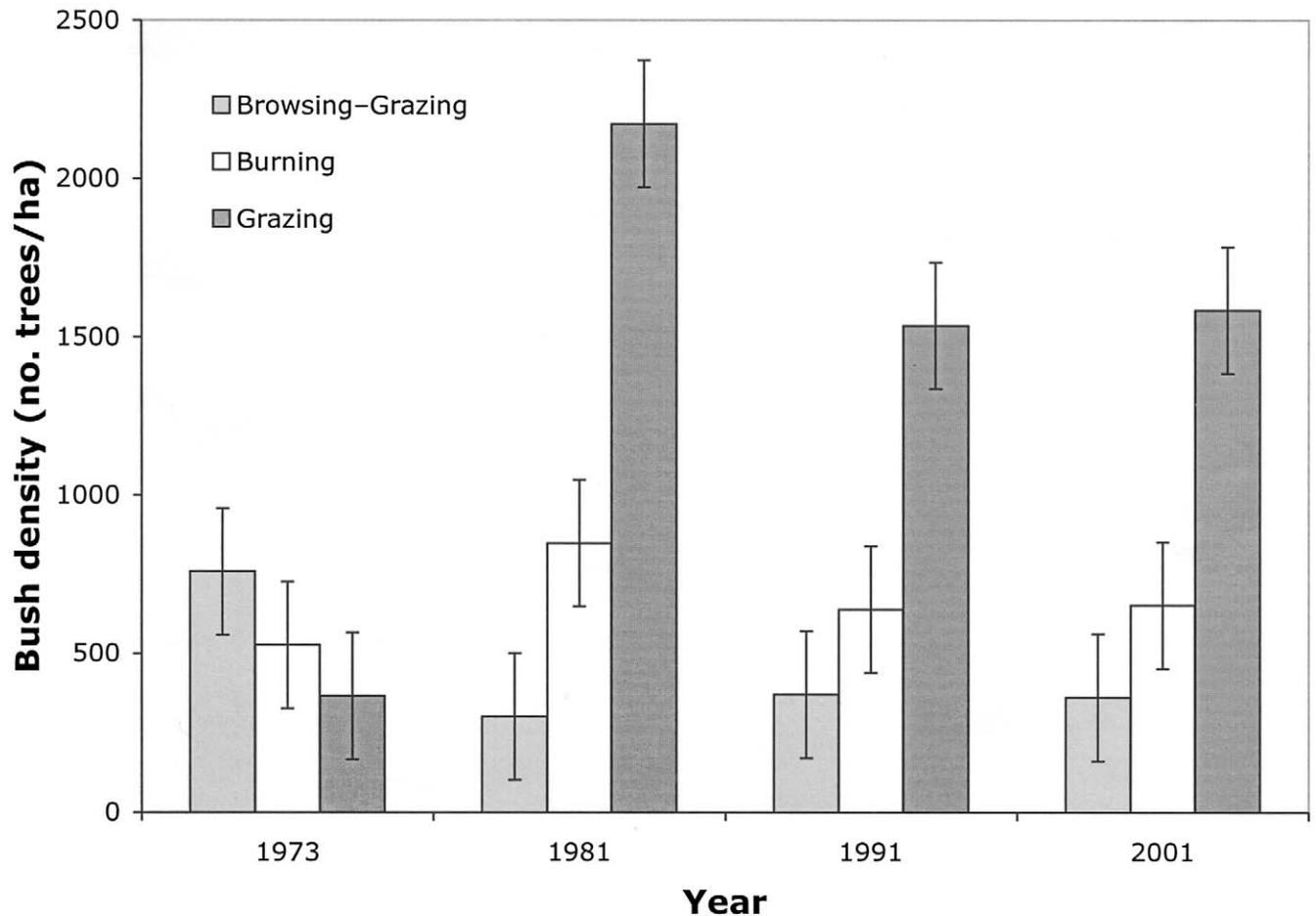


Fig. 4. Effect of grazing with cattle, browsing with goats, and annual burning on tree density (no. trees/ha) in the False Thornveld of the Eastern Cape, South Africa, 1973–2001.

1973 there were no marked differences in bush density in the different treatments. However, by 1981 distinct differences had developed and the browsing–grazing treatment had resulted in a significant reduction in density of trees and shrubs relative to 1973, and this was maintained up until the assessment in 2001 (29 y). Conversely, the grazing and annual burning treatments resulted in significant increases in density of trees and shrubs, with the grazing treatment resulting in the highest overall density of bush (Trollope and Dondofema 2003). This example illustrates the importance of the interacting effects of fire and herbivory on savanna vegetation when used as a range management practice in Africa.

FIRE MANAGEMENT

Research on the effects and use of fire in southern and East African grasslands and savannas (Trollope 1983, 1989; Trollope and Potgieter 1986; van Wilgen et al. 1990; Trollope and Trollope 1999; Trollope et al. 2000, 2003) has led to the conclusion that grass, trees, and shrubs in various localities in Africa react similarly to different components of the fire regime and, therefore, general guidelines can be formulated for prescribed burning. However, for the sake of clarity,

guidelines for prescribed burning will be dealt with separately for the use of fire as a range management for domestic livestock husbandry and for wildlife management as the broad management objectives vary for these different forms of land use.

PRESCRIBED BURNING FOR DOMESTIC LIVESTOCK SYSTEMS

Prescribed burning is an important and often essential range management practice in areas used for either commercial or subsistence livestock farming. The most important factors to consider when planning a burning program are the reasons for burning and the appropriate fire regime to be applied.

Reasons for Prescribed Burning

The consensus among range scientists and progressive livestock farmers on permissible reasons for prescribed burning are 1) to remove moribund (i.e., leaf and stem material on grass plants that are overgrown and suffering from self-shading) and/or unacceptable (i.e., leaf and stem material on grass plants that is unpalatable to grazing animals) grass material, and 2) to control and/or prevent the encroachment of

undesirable plants (Trollope 1989). These basic reasons for burning grassland and savanna vegetation in Africa are equally applicable to commercial or subsistence livestock farming.

An often quoted reason for burning rangeland is to stimulate an out-of-season "green bite." This is often done during summer, late autumn, or late winter to provide green nutritious regrowth for grazing by livestock. This practice is unacceptable because 1) it reduces vigor of the grass sward, 2) it reduces basal and canopy cover of the grass sward, 3) it increases runoff of rainwater, and 4) it can result in accelerated soil erosion. This malpractice cannot be sufficiently condemned as it has been responsible for drastic deterioration in range condition over extensive areas of southern Africa, and unfortunate examples are evident at numerous sites in the eastern mountain grasslands of South Africa.

It has been suggested that fire can also be used to control ticks that cause tick-borne diseases in livestock, but this reason is generally discounted because ticks persist in areas that are frequently burned. However, Stampa (1959), in a study of the Karoo paralysis tick (*Ixodes rubicundus*) in the Karroid *Merxmuellera* Mountain Veld in South Africa, has shown that this parasite can be successfully controlled by altering the microclimate at the soil level and thereby creating an unfavorable habitat for this organism, resulting in its disappearance. Similar evidence has been obtained by Trollope et al. (2003) in the Ngorongoro Crater and Serengeti grasslands in Tanzania, where controlled burning by nomadic Masai pastoralists has resulted in a significantly lower incidence of ticks where this practice is applied. This latter finding has resulted in the reintroduction of prescribed burning in the Ngorongoro Crater, with beneficial results.

Finally it was shown that frequent fires favor the abundance of the productive and palatable grass species *Themeda triandra* in southern African grasslands (Scott 1971, Dillon 1980, Forbes and Trollope 1991). This raises the possibility of using fire to improve range condition by increasing the abundance of valuable forage species like *Themeda triandra*. Following appropriate research on the response of key forage species to fire, forage condition improvement could be considered as a valid reason for burning rangeland in the future.

Ecological Criteria for Prescribed Burning

The necessity for burning rangeland depends upon its ecological status and physical condition. Generally the condition of the grass sward determines whether rangeland should be considered for burning. This component reflects the ecological status of the ecosystem and the presence of or its ability to produce adequate grass fuel to carry and support a fire. Quantitative techniques have been developed in southern and East Africa to assess condition of the grass sward in relation to the need for prescribed burning (Trollope and Potgieter 1986, Hardy et al. 1999, Trollope and Trollope 1999, Trollope et al. 2000). The condition of the grass

sward must be determined in terms of its botanical composition, ecological status, and basal cover, and involves classifying different grass species into different ecological categories according to their response to a grazing gradient, from high to low grazing intensities:

- 1) Decreaser species: Grass and herbaceous species that decrease when rangeland is under- or overgrazed;
- 2) Increaser I species: Grass and herbaceous species that increase when rangeland is under- or selectively grazed;
- 3) Increaser II species: Grass and herbaceous species that increase when rangeland is overgrazed.

Prescribed burning should not be applied if the grass sward is in a pioneer condition dominated by Increaser II grass species caused by overgrazing. Burning is generally not recommended when rangeland is in this condition in order to enable it to develop to a more productive stage dominated by Decreaser grass species. Conversely, when the grass sward is in an undergrazed condition dominated by Increaser I species, it needs to be burned to increase the fire-adapted and more productive Decreaser grass species. Finally, prescribed burning is necessary when the grass sward has become overgrown and moribund as a result of excessive self-shading. These conditions develop when the standing crop of grass is generally $\geq 4,000$ kg/ha and can be estimated with the Disc Pasture Meter developed by Bransby and Tainton (1977).

The criteria used for deciding whether to burn to control or prevent the encroachment of undesirable plants are the same criteria describing the condition of the grass sward. However, grass fuel loads required for prescribed burning for this reason may differ depending on the encroaching plant species.

Fire Regime

In this discussion, the fire regime to be used in prescribed burning refers to the type and intensity of fire and the season and frequency of burning.

Type of Fire

It is recommended that fires burning with the wind either as surface headfires in grassland or a combination of surface headfires and crown fires in tree and shrub vegetation be used in prescribed burning because they cause least damage to the grass sward but can cause maximum damage to woody vegetation if required (Trollope 1999).

Fire Intensity

Research on fire behavior in Eastern Cape Province and Kruger National Park has shown that fire can be classified into categories according to fire intensity (Table 3) (Trollope 1983, Trollope and Potgieter 1985). When burning to remove moribund and/or unacceptable grass material, a cool fire of $< 1,000$ kJ s⁻¹ m⁻¹ is recommended. This can be achieved by burning

Table 3. Fire classification according to fire intensity.

Fire intensity (kJ s ⁻¹ m ⁻¹)	Description
<500	Very cool
501–1,000	Cool
1,001–2,000	Moderately hot
2,001–3,000	Hot
>3,000	Extremely hot

when the air temperature is <20°C and the relative humidity is >50%. When burning to control undesirable plants like encroaching bush, a hot fire of >2,000 kJ s⁻¹ m⁻¹ is necessary. This can be achieved when the grass fuel load is >4,000 kg/ha, the air temperature is >25°C, and the relative humidity is <30%. A hot fire will cause a significant top-kill of stems and branches of bush species up to a height of 3 m. In all cases, the wind speed should not exceed 20 km/h.

Season of Burning

Research in southern Africa has clearly indicated that damage to the grass sward is minimized if burning is applied when the grass is dormant. Therefore, it is recommended that when burning to remove moribund and/or unacceptable grass material, burning should preferably be applied after the first rains of >13 mm at the commencement of the growing season, i.e., when grass is still dormant and fire hazard is low. Conversely, when burning to control encroaching plants, burning should be applied before the first rains of the growing season, i.e., when grass is very dry and dormant to ensure a high-intensity fire. The actual time of the year when prescribed burning will be applied in Africa depends on the latitude and rainfall pattern of the region.

Frequency of Burning

When burning to remove moribund and/or unacceptable grass material, frequency of burning will depend upon the rate of accumulation of excess grass litter (Trollope 1999). Field experience indicates that litter accumulation should not exceed 4,000 kg/ha and, therefore, frequency of burning should be based on the rate at which this phytomass of grass material accumulates. This approach has the advantage that the frequency of burning is related to the stocking rate of grazers and to the amount of rainfall the area receives. Generally in high rainfall areas (>700 mm/y) this will dictate a frequency of burning every 2–4 y. In more arid areas, the frequency will be much lower and, in fact, the threshold of a grass fuel load >4,000 kg/ha will generally exclude fire in these regions, particularly where condition of the rangeland is degraded and excessive grass fuel loads never accumulate.

Post-Fire Range Management

When burning to remove moribund and/or unacceptable grass material, grazing should be applied as soon as possible after the burn to take advantage of the highly nutritious regrowth of grass. It is unclear

whether rotational or continuous grazing should be applied after fire. However, there is complete consensus among rangeland scientists on the necessity of applying a rotational resting system when prescribed burning is used (Zacharias 1994, Kirkman 2002). This involves withdrawing a portion of the rangeland from grazing for an extended period of at least 1 growing season or longer (6–12 mo) to maintain the vigor of grasses and enable seed production to occur for plant recruitment. The rest period is applied during the season prior to the prescribed burn. In terms of rotational grazing after a burn, great success has been obtained with the “open camp system” developed in KwaZulu-Natal. Grazing camps are permanently established fenced areas or paddocks on a commercial livestock ranch, i.e., the rangeland on a livestock ranch is subdivided into grazing camps that are grazed on a rotational basis. This system involves burning a camp and grazing it as soon as possible after the fire, after which the livestock are moved rotationally to other camps until such time as the burned camp is ready to be grazed again. Burned rangeland is thus maintained in a palatable and nutritious condition for as long as possible after the burn to the benefit of livestock. The same procedure is then followed in subsequent years. This system presupposes the availability of adequate camps to apply this form of grazing management. In areas where there are few grazing camps available, emphasis must be given to applying a rotational resting system. Where there are no camps, such as in communal grazing areas like the Transkei in South Africa, sufficiently large areas need to be burned to avoid overutilizing the burned area. This practice will also de facto result in a resting treatment being applied to the unburned area, which is initially less attractive to grazing animals.

PRESCRIBED BURNING FOR WILDLIFE MANAGEMENT

Prescribed burning is recognized as an important management practice for wildlife management in African grassland and savanna ecosystems and is regarded as a natural factor of the environment essential for the ecological well-being of both biotic and abiotic components (Trollope 1990, Thomson 1992, Bothma 1996). However, a wide diversity of views prevails on the most appropriate burning system to use in wildlife areas, ranging from so-called “natural” burning systems based entirely on lightning as the ignition source to actively applied burning systems based on rangeland condition.

Fire management systems are most developed in southern Africa. The different burning systems used are the lightning burning system, the range condition burning system, the patch mosaic burning system, and the integrated fire management system. Personal experience in southern and East African grasslands and savannas has led to the conclusion that the range condition burning system is a practical and efficient burn-

ing system to use in wildlife areas and is strongly recommended.

Range Condition Burning System

The range condition burning system was developed from a fire research program that was initiated in Eastern Cape Province in 1968 and later extended to Kruger National Park in 1982 and to East Africa in 1992 (Trollope 1971, Trollope and Potgieter 1985, Trollope and Trollope 1999). It is based on empirical results and is appropriate for use in wildlife areas because it provides a practical means of improving and maintaining species and habitat diversity of natural grassland and savanna ecosystems (Trollope 1971, Trollope et al. 1995). It can be used to remove moribund and/or unacceptable grass material, to create or maintain an optimum relationship between herbaceous and woody vegetation, if necessary, and to encourage wildlife to move to less preferred areas in order to minimize the overutilization of preferred areas. The basic philosophy of the range condition burning system is that the use of fire to achieve specific management objectives must be based on the condition of the vegetation and its known response to different components of the fire regime. Similar ecological criteria, as used for prescribed burning for domestic livestock systems, are also used to select areas for burning. This system also allows for all wildfires initiated by unplanned ignition sources, such as lightning or other causes, to burn freely if the grass sward is dominated by Decreaser and/or Increaser I grass species and is in a moribund and unpalatable condition. The effect of fire on grassland and savanna vegetation is similar irrespective of the source of ignition when burned under the same weather conditions. Finally, limits must be set on the extent of area to be burned at any one time as a precaution against inadequate forage being available for herbivorous wildlife. It is therefore recommended that not >50% of a given area be burned in moist grassland and savanna ecosystems (>700 mm rainfall/y) and not >33% in arid (<500 mm rainfall/y) grassland and savanna ecosystems.

The designated areas to be burned can be ignited either as point ignitions or as perimeter ignitions involving "block burning." Point ignitions are used to develop a mosaic of different types of fires in response to changes in wind direction, air temperature, and relative humidity during the burning period. The ensuing fire mosaic will ensure a range of fire effects, which are intended to maximize habitat diversity both in grassland and woody components of the vegetation. The use of point ignitions is best suited to large conservation areas generally >20,000 ha (Brockett et al. 2001), where intensive range management is not a critical requirement. The intensity of prescribed burns will vary according to the objectives for burning similar to domestic livestock systems. Regarding season of burning it is recommended that prescribed burns be applied when the grass sward is dormant, but the "burning window" can extend over the entire dry season and the actual timing of fires may be varied according to

objectives for burning. When burning to remove moribund grass material, prescribed burns can be applied at any time during the dormant season. When burning to reduce the density and size of trees and shrubs, fires should be applied when grass fuel is at its lowest moisture content in order to ensure a high-intensity fire. Finally, the guidelines used for controlling the frequency of burning in domestic livestock systems also apply to prescribed burning for wildlife management.

Post-Fire Range Management

Grazing after burning in wildlife areas is difficult to control. In order to prevent overgrazing, it is important to ensure that the burned area exceeds the short-term forage requirements of grazing animals that are attracted to the highly palatable and nutritious regrowth that develops after a burn (Trollope 1992). Another strategy that has been successful in southern Africa is to apply a series of patch burns at regular intervals throughout the duration of the burning window during the dormant season. This attracts grazing animals to successively burned areas, thereby spreading the impact of grazing over the entire burned area and avoiding the detrimental effects of heavy continuous grazing in any one area (Brockett et al. 2001).

GENERAL DISCUSSION AND CONCLUSIONS

An important feature of this overview of the fire ecology of African grasslands and savannas is the dependence of prescribed burning on a knowledge of fire behavior and fire effects. An attempt has been made to quantify primary factors affecting fire behavior in terms of rate of spread, fire intensity, and flame height of surface headfires. The influence of various factors contributing to fire intensity in African grasslands and savannas has also been quantified. Significant progress has been made with determining the effects of type and intensity of fires on the grass sward and tree and shrub vegetation in African grass and savanna ecosystems. This research has enabled formulating effective and practical guidelines for fire as a range management practice for domestic livestock production and wildlife management. However, research conducted to develop the fire intensity model and to determine the effects of type and intensity of fires on herbaceous and woody vegetation has been largely limited to savanna areas of the Eastern Cape Province and Kruger National Park. This emphasizes the need to extend similar fire ecological studies to other African grassland and savanna areas. The greatest limiting factors to conducting fire ecology studies are to find suitable study sites where landowners and users are prepared to sacrifice forage supplies that are normally used for sustaining either domestic livestock or wildlife populations. In addition, applying controlled experimental fires is dangerous, requiring well-developed managerial skills and suitable firefighting equipment to contain fires, all of which are not readily available over much of the African continent. Finally, competent fire ecologists are

a rare commodity, not only in Africa but in the world, further limiting the implementation of a comprehensive fire research program elsewhere in Africa. The most successful attempt to test these results elsewhere in Africa has been to conduct fire behavior and fire effect trials in the Kenya central highlands. These trials lead to the conclusion that the behavior and effect of surface fires in East African grasslands and savannas were similar to that in southern African grasslands and savannas (Trollope and Trollope 1999).

Another successful attempt was to compare the behavior of fires in African grasslands and savannas with fires occurring in tallgrass prairies on the Konza Prairie Biological Station at Kansas State University in the United States. This involved recording the behavior of 45 surface headfires and 24 surface backfires applied as controlled burns to catchment units on Konza. A detailed analysis of the fire behavior data led to the conclusion that the overall behavior of surface headfires in tallgrass prairies at Konza was similar to that in grasslands and savanna ecosystems in South Africa (Trollope et al. 2004). These two independent data sets from similar types of vegetation, but in widely separated grassland and savanna areas on the earth, provide a measure of scientific support for research results that have been obtained on fire behavior and to an extent on fire effects in southern African grasslands and savannas. Nevertheless, it is strongly recommended that as much further research as possible be conducted in other African grassland and savanna ecosystems. This will contribute to a greater understanding of the fire ecology of these ecosystems, thereby leading to improved range management for both domestic livestock production and wildlife management on the African continent.

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A CONTRAST IN SIMILARITIES: FIRE AND PLANT DIVERSITY IN GRASSLANDS OF NORTH AMERICA AND WESTERN AUSTRALIA

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ABSTRACT

The tallgrass prairie (*Andropogon-Panicum*) of central North America and the spinifex (*Triodia*) grasslands of the Gibson Desert of central Western Australia differ substantially in many ways, including in their responses to fire, but are similar in that both confirm the importance of fire in maintaining ecosystem diversity. In tallgrass prairie, the historic fire return interval is estimated to have averaged 3–5 y, most likely dominated by large-scale dormant-season fires with more frequent but smaller growing-season fires. Fires were initiated by both Native Americans and lightning, with effects most likely heterogeneous due to factors such as patchy grazing of large grazers that altered fuel continuity. Fire in this ecosystem prevented extensive woody plant invasion and maintained a diverse plant community by invigorating the perennial prairie plants and by facilitating seedling germination and establishment. In the spinifex grassland of the Gibson Desert of central Western Australia, an ecosystem interspersed with bands of mulga (*Acacia aneura*), the frequency of historic fires is not as well known. Aboriginal and lightning-caused fires are reported to have maintained much of the landscape as a patchwork of different post-fire successional states presumably supporting a diverse flora and fauna. In contrast to the tallgrass prairie, grazers are not likely to have influenced fire heterogeneity because the dominant grass (*Triodia basedowii*), which is highly flammable even when green, appears unpalatable to any organism except termites. Further, in contrast to the post-fire resprouting of perennial plants of the tallgrass prairie, *T. basedowii* and most other plants of the spinifex grassland do not survive fire, instead regenerating from seeds dependent largely on post-fire rainfall. Thus, fuel accumulation and continuity in the spinifex grasslands, and consequently fire frequency, would have been dictated mostly by post-burn growing conditions. In addition, plant diversity in the spinifex grasslands results from the release of buried seed following fire-caused death of *T. basedowii* hummocks. The dramatic post-fire increase in plant diversity, which declines to pre-burn conditions within 5 y, is in sharp contrast to the maintenance of diversity afforded by fire in the tallgrass prairie. Thus, the means by which diversity is maintained in the tallgrass and spinifex grasslands differs substantially—maintenance versus cyclic replacement—but in both types of grasslands, fire is the central player. Not unexpectedly, the dynamics of both ecosystems are threatened by alteration of the historic fire regimes, with one consequence an apparent decline in plant (and perhaps animal) diversity.

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ARE LIGHTNING FIRES UNNATURAL? A COMPARISON OF ABORIGINAL AND LIGHTNING IGNITION RATES IN THE UNITED STATES

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ABSTRACT

It is now widely acknowledged that frequent, low-intensity fires once structured many plant communities. Despite an abundance of ethnographic evidence, however, as well as a growing body of ecological data, many professionals still tend to minimize the importance of aboriginal burning compared to that of lightning-caused fires. Based on fire occurrence data (1970–2002) provided by the National Interagency Fire Center, I calculated the number of lightning fires/million acres (400,000 ha) per year for every national forest in the United States. Those values range from a low of <1 lightning-caused fire/400,000 ha per year for eastern deciduous forests, to a high of 158 lightning-caused fires/400,000 ha per year in western pine forests. Those data can then be compared with potential aboriginal ignition rates based on estimates of native populations and the number of fires set by each individual per year. Using the lowest published estimate of native people in the United States and Canada prior to European influences (2 million) and assuming that each individual started only 1 fire per year—potential aboriginal ignition rates were 2.7–350 times greater than current lightning ignition rates. Using more realistic estimates of native populations, as well as the number of fires each person started per year, potential aboriginal ignition rates were 270–35,000 times greater than known lightning ignition rates. Thus, lightning-caused fires may have been largely irrelevant for at least the last 10,000 y. Instead, the dominant ecological force likely has been aboriginal burning.

keywords: aboriginal burning, Indian burning, lightning-caused fires, lightning-fire ignition rates, potential aboriginal ignition rates.

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INTRODUCTION

It is now widely acknowledged that frequent, low-intensity fires once structured many plant communities in the United States. Anderson (2005), Stewart (1956, 1963, 2002), Zybach (2003), Lewis (1973, 1977, 1985), Pyne (1982, 1993, 1994, 1995), and others (Blackburn and Anderson 1993, Kay and Simmons 2002, Carloni 2005, Gassaway 2005) contend that, historically, most fires were set by native people to manage their environment. Vale (2002), Baker (2002), and their colleagues (Houston 1973, Loope and Gruell 1973), however, maintain that the case for aboriginal burning has been overstated and that most fires, historically, were started by lightning. According to Baker (2002:41–42), “Ignitions by Indians were . . . probably numerically insignificant relative to lightning ignitions . . . [and] Indians were a small part of a large Rocky Mountain wilderness, with a fire regime . . . essentially free of human influence for millennia.” However, neither Vale (2002) nor Baker (2002) presented data on actual lightning ignition rates nor compared known lightning ignition rates with potential aboriginal ignition rates. In this paper, I present data on lightning-fire ignition rates for every national forest in the contiguous United States and then compare those figures with potential aboriginal ignition rates based on hypothetical estimates of native populations and the

number of fires accidentally and purposefully set by each individual per year.

LIGHTNING-FIRE IGNITION RATES

The National Interagency Fire Center in Boise, Idaho, provided data on the number of known lightning-caused fires that occurred on individual national forests from 1970 to 2002. Based on the area of each forest, I then calculated lightning-fire ignition rates/million acres (400,000 ha) per year (Table 1). Those data range from a low of <1 fire/400,000 ha per year to 158 fires/400,000 ha per year on the Plumas National Forest in California. Ponderosa pine (*Pinus ponderosa*)–dominated forests in Arizona and New Mexico also have high lightning-fire ignition rates but, surprisingly, most national forests have relatively low lightning-fire ignition rates—this is especially true of national forests in the East (Figures 1, 2). Even the majority of western national forests, though, have relatively low lightning ignition rates (Figures 1, 2). Several national forests in Montana, Wyoming, and Colorado have <10 lightning-caused fires/400,000 ha per year (Table 1). National forests also have higher lightning-fire ignition rates than surrounding, lower-elevation, Bureau of Land Management (BLM), state, and private land (Barrows 1978). When those data are included, the mean lightning-fire ignition rate on all lands in the western United States is approximately 19 fires/400,000 ha per year (Table 2).

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These data then do not support the idea that the United States, or even the West, is awash in lightning-started fires. Popular misconceptions regarding the frequency of lightning fires may be due to media coverage during recent extreme fire seasons, as well as the fact that many fire-history studies have been done on the few national forests in California, Arizona, and New Mexico that have relatively high lightning-fire ignition rates.

POTENTIAL ABORIGINAL IGNITION RATES

Any estimate of aboriginal ignition rates must consider at least three factors—the number of landscape fires started inadvertently per person per year, the number of fires purposefully set per person per year, and the number of people. Unfortunately, how many people there were in the Americas prior to Columbus' landfall is not a settled issue. In fact, the entire subject is exceedingly contentious and highly charged, as it impinges directly on various national creation beliefs, charges of genocide by remaining indigenous inhabitants, and core environmental values, such as the idea of wilderness (Stannard 1992, 1998; Loewen 1995; Churchill 1997; Kay and Simmons 2002; Vale 2002; Mann 2005). Then, too, there is the problem that European-introduced diseases, such as smallpox, decimated native populations well in advance of actual European contact.

Smallpox, to which Native Americans had no acquired or genetic immunity, entered the Americas around 1520 and, according to Dobyns (1983), native people attempting to escape Spanish domination in Cuba fled to Florida in ocean-going canoes and brought smallpox to the mainland. Dobyns postulated that at least three major pandemics swept North America and reduced aboriginal populations by 90% or more before the Pilgrims arrived at Plymouth Rock. Needless to say, Dobyns' hypothesis has caused a great deal of debate, but recent archaeological work by Ramenofsky (1987), Campbell (1990), and Kornfield (1994) has documented a major aboriginal population collapse in the northern Rockies and on the northern Great Plains ca. 1550–250 y before explorers like Lewis and Clark (1804–1806) set foot in the West. Thus, we are left with a range of estimates—from a low of only a few million aboriginal inhabitants to a high of 200–300 million in the Americas ca. 1491 (Mann 2005). The only certainty is that Europeans have consistently underestimated the antiquity of aboriginal occupation, as well as the political and technical sophistication of America's original inhabitants (Mann 2005).

To be conservative in my evaluation of potential aboriginal ignition rates, I started with the lowest, published and commonly accepted estimate that I could find, namely 2 million native people in the continental United States and Canada ca. 1491 (Mann 2005). As there are approximately 1.5 billion ha north of Mexico, this yields a density estimate of 428 people/400,000

ha. Assuming there were only 500,000 natives in that area, as Alroy (2001) calculated for the end of the Pleistocene, then the density estimate is 107 people/400,000 ha. Both seemingly insignificant figures.

Escaped Campfires—Inadvertent Landscape Burning

Another thing that can be stated with certainty is that no one has ever found a Smokey Bear poster in an archaeological site anywhere in North or South America. In fact, no evidence exists that native people ever purposefully extinguished their heating or cooking fires. Most likely, they simply walked away and left their campfires burning.

In a very extensive search of the literature, I discovered almost no reference that natives anywhere carefully extinguished fires. . . . Everywhere that man traveled, he made campfires and left them to ignite any and all vegetation in the vicinity [Stewart 1956:118].

If native people routinely used water or soil to put out campfires, we would expect to find large pieces of charcoal in archaeologically recovered fire pits, but charcoal is rare or absent from such features—all that is commonly found is white ash or exceedingly fine charcoal particles. Wright (1984:20–21), who conducted extensive archaeological research in the Yellowstone ecosystem, noted:

We have recorded nearly three dozen archaeological sites spanning about 4000 years of occupation. Rock broken from the heat of campfires is abundant, but charcoal is virtually absent. Even though it requires only four grams of charcoal for a C-14 analysis, on not one site has enough been collected for a date. There is obvious evidence of extensive cooking, so what has happened to the burned wood? At Blacktail Butte the firepits were shallow and the wind blows hard. No doubt much of the charcoal was dispersed by the wind, quite probably as still burning embers. The chance of accidental fires was quite high.

The only cases in which large pieces of charcoal have routinely been unearthed in archaeological settings are where habitation structures were set on fire, and this is usually interpreted as a sign of conflict or warfare (William Hildebrandt, Far Western Anthropological Research Group, personal communication).

Similarly, anthropologists who work with modern-day hunter-gatherers living in South America, Australia, and Africa report that their subjects never extinguish heating or cooking fires unless under duress by Europeans (Jim O'Connell, University of Utah, personal communication; William Preston, California Polytechnic State University, personal communication; Richard Chacon, Winthrop University, personal communication). Peter Fidler, who traveled with a band of Piegan natives in what is today central and southern Alberta during the winter of 1792–1793, reported how

Table 1. Lightning-fire ignition rates on national forest lands in the United States. Fire occurrence data (1970–2002) provided by the National Interagency Fire Center, Boise, ID.

National forest	Number of lightning fires/400,000 ha per year
Western United States	
Arizona	
Apache-Sitgreaves	81
Coconino	150
Coronado	49
Kaibab	97
Prescott	43
Tonto	61
California	
Angeles	26
Cleveland	17
Eldorado	49
Inyo	31
Klamath	64
Lassen	52
Los Padres	8
Mendocino	23
Modoc	51
Plumas	158
San Bernardino	121
Sequoia	75
Shasta-Trinity	38
Sierra	65
Six Rivers	18
Stanislaus	57
Tahoe	56
Colorado	
Arapaho-Roosevelt	12
Grand Mesa–Uncompahgre–Gunnison	8
Pike–San Isabel	25
Rio Grande	5
Routt	7
San Juan	32
White River	7
Idaho	
Boise	47
Caribou	14
Challis	16
Clearwater	70
Nez Perce	65
Panhandle	27
Payette	49
Salmon	31
Sawtooth	12
Targhee	12
Montana	
Beaverhead	8
Bitterroot	65
Custer	46
Deerlodge	13
Flathead	16
Gallatin	8
Helena	21
Kootenai	39
Lewis and Clark	9
Lolo	45
Nebraska	
Nebraska	73
Nevada	
Humboldt	7
Toiyabe	25
New Mexico	
Carson	22
Cibola	38
Gila	105
Lincoln	35
Santa Fe	55

Table 1. Continued.

National forest	Number of lightning fires/400,000 ha per year
Oregon	
Deschutes	54
Fremont	43
Malheur	83
Mount Hood	20
Ochoco	79
Rogue River	68
Siskiyou	14
Siuslaw	1
Umpqua	59
Wallowa–Whitman	50
Willamette	43
Winema	45
Umatilla	59
South Dakota	
Black Hills	64
Utah	
Ashley	22
Dixie	34
Fishlake	28
Manti-La Sal	33
Uinta	16
Wasatch-Cache	10
Washington	
Gifford Pinchot	14
Mount Baker–Snoqualmie	7
Okanogan	35
Olympic	6
Wenatchee	27
Wyoming	
Bighorn	8
Bridger–Teton	11
Medicine Bow	18
Shoshone	6
Eastern United States	
Alabama	
All national forests	6
Arkansas	
Ouachita	9
Ozark–St. Francis	4
Florida	
All national forests	51
Georgia	
Chattahoochee–Oconee	3
Illinois	
Shawnee	0.3
Kentucky	
Daniel Boone	1
Louisiana	
Kisatchie	2
Michigan	
Hiawatha	1
Huron–Manistee	1
Ottawa	1
Minnesota	
Chippewa	1
Superior	6
Mississippi	
All national forests	1
Missouri	
Mark Twain	1
New Hampshire	
White Mountain	1
North Carolina	
All national forests	2

Table 1. Continued.

National forest	Number of lightning fires/400,000 ha per year
Ohio–Indiana	
Wayne–Hoosier	0.1
Pennsylvania	
Allegheny	0.1
South Carolina	
Sumter–Francis Marion	3
Tennessee	4
Cherokee	
Texas	
All national forests	3
Vermont	
Green Mountain	0.3
Virginia	
George Washington–Jefferson	2
West Virginia	
Monongahela	0.4
Wisconsin	
Chequamegon	1
Nicolet	1

aboriginal attitudes toward fire differed from those of Europeans:

2 Tents [of Piegan] joined us that was tenting ¾ mile to the Eastward of us. They did not put out their fire when they left it, which spread amongst the dry grass and ran with great velocity and burnt with very great fury, which enlightened the night like day, and appeared awfully grand. The wind being fresh drove it at a great distance in a little while [Haig 1992:58].

This observation was recorded on 18 January, a time of year when lightning-started fires are nonexistent on the northern Great Plains (Higgins 1984).

So, to begin with a simple and conservative assumption that there was only 1 escaped campfire/y per adult aboriginal inhabitant, and using the previous estimate of 428 native people/400,000 ha, this produces an estimate of 428 escaped fires/400,000 ha per year, which is 2.7 times the highest known lightning ignition rate in the West or 350 times the lightning ignition rate for national forests in the East (Table 1). If, on the other hand, we assume there were 20 million native inhabitants, possibly a more realistic figure (Dobyns 1983, Mann 2005), then the estimated escaped-campfire ignition rate is 27 times higher than the highest known lightning ignition rate and 3,500 times higher than the lightning ignition rate in much of the eastern United States. If we assume 10 escaped campfires/y per aboriginal inhabitant, instead of 1, then the accidental ignition rate is 270 times the highest lightning-started rate and 35,000 times the lightning-fire ignition rate in the East.

Unlike Europeans, aboriginal people without metal cutting instruments, which included all the Americas before 1492, tended to build relatively small cooking and warming fires. First, it took work to collect the necessary firewood and second, because large fires were more likely to be detected by enemies. Thus, no

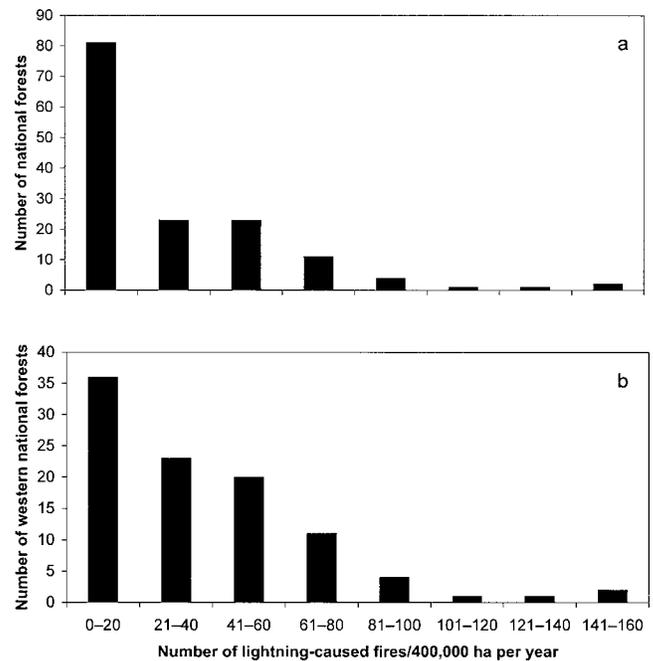


Fig. 1. Lightning-fire ignition rates on national forests in the contiguous United States (not including national grasslands). (a) All national forests (not including Alaska). (b) National forests west of the 100th Meridian (not including the Chugach and Tongass national forests, Alaska).

more than 6–8 native people usually sat around a single campfire (Binford 1978, Kelly 1995, Hill and Hurtado 1996). Assuming that 8 people shared a single campfire, that there were 2 million aboriginal inhabitants north of Mexico, and that each group of 8 lit only 1 campfire/d, this calculates out to 19,500 fires/400,000 ha per year—all of which were presumably left burning. This is 124 times the highest known lightning ignition rate (Table 1). However, it should be noted that some large villages of native peoples did occur in the East in the 1500s and were associated with extensive agriculture, such as near present-day Tallahassee in North Florida (Masters et al. 2003). This would likely decrease the potential for escapes in our hypothetical example.

Baker (2002:41) dismissed aboriginal burning as a significant ecological force, in part because he contended that “only about 30,000” native people inhabited the northern Rockies. Baker did not define what he considered the northern Rockies but if we assume this includes one-half of Colorado, one-half of Montana, one-half of Wyoming, and one-third each of Idaho and Utah, we have an area of 610,000 km² (235,000 mi²) or 1 aboriginal inhabitant/19 km² (7.3 mi²). Again, a seemingly insignificant figure. A number, however, that translates to 212 people/400,000 ha. The mean lightning ignition rate for national forests in the northern Rockies, though, is only 17.6 fires/400,000 ha per year (Table 1). Assuming only 1 escaped campfire/aboriginal person per year, the accidental ignition rate is still 12 times the lightning ignition rate. Any other assumptions, as to the number of escaped campfires, only put more fire on the landscape. Thus, this hypothetical example does not support Baker’s (2002:

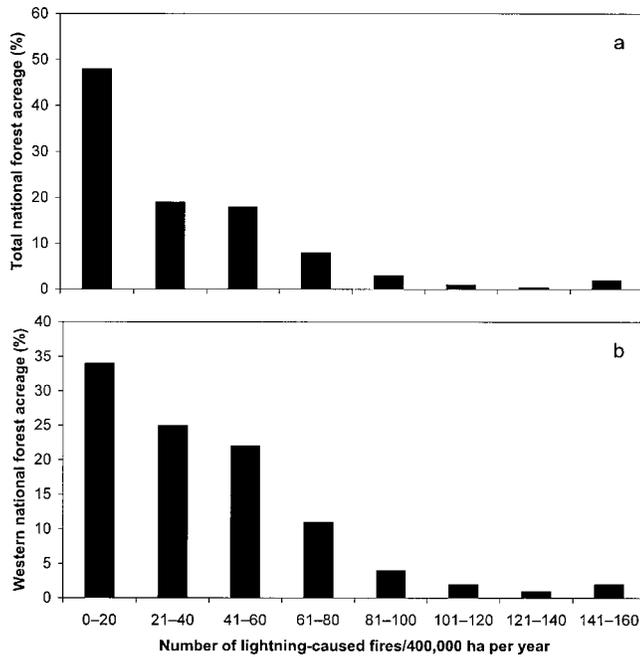


Fig. 2. Lightning-fire ignition rates by area for national forests in the contiguous United States (not including national grasslands). (a) All national forests. (b) National forests west of the 100th Meridian (not including the Chugach and Tongass national forests, Alaska).

41) conclusions that aboriginal fires were “insignificant” or that the Rockies were a wilderness untouched by the hand of man.

Similarly, Griffin (2002:81) suggested that there may have been no more than 1 native person/23 km² (8.9 mi²) in the Great Basin and therefore aboriginal burning was unimportant compared to lightning-started fires. Griffin’s aboriginal population estimate translates to 176 people/400,000 ha. For comparison, national forests in Nevada have a lightning ignition rate of only 17.8 fires/400,000 ha per year (Table 1). Using the conservative assumption of 1 escaped campfire/person per year, the accidental aboriginal ignition rate was 10 times the known lightning ignition rate. Thus, available data suggest that accidentally started aboriginal fires were 1, 2, or 3 orders of magnitude greater than known lightning ignition rates in the United States—depending on location and vegetation type (Fechner and Barrows 1976:19). For other reviews of the methodology used by Vale (2002), Baker (2002), and Griffin (2002), see LaLande (2003) and Pyne (2003). Finally, despite an extremely successful anti-fire public relations campaign, fire bans, and other measures, including closing entire national forests during high fire danger, 49% of the fires recorded in the National Forest System from 1940 to 2000 were caused by humans, not by lightning—and those human-set fires accounted for 57% of the area burned (Stephens 2005).

Purposeful Burning—Management-Set Fires

Although there is little doubt that Native Americans used fire to purposefully modify their environ-

Table 2. Average lightning-fire ignition rates on protected state, private, and federal lands in the western United States, 1960–1975 (Barrows 1978:4).

State	Number of lightning fires/400,000 ha per year
Arizona	46
New Mexico	21
Colorado	11
Wyoming	6
Idaho	25
Montana	17
Nevada	3
Utah	7
California	28
Oregon	30
Washington	19
All western states	19

ment (Stewart 1963, 2002; Lewis 1973, 1977, 1985; Anderson 2005), ethnographers have failed to record the number of fires set/person per year. The only data that I have been able to locate on this subject come from Australia where, in a few locations, aboriginal people still use fire to purposefully modify the vegetation as their ancestors are thought to have done for the last 45,000–50,000 y (Hallam 1975, Lewis 1989, Flannery 1994, Fensham 1997, Russell-Smith et al. 1997, Bowman 1998, Bowman et al. 2004, Vigilante and Bowman 2004). In Australia, most of the aboriginal-set management fires are started by men and each individual sets 100 or more fires/y, mostly at the end of the wet season and the beginning of the dry season—a time when lightning-fires are rare to nonexistent. This creates a vegetation mosaic that not only is more productive for the indigenous inhabitants but which also prevents large-scale, high-intensity, lightning-caused fires during the height of the dry season. Aboriginal-managed areas have also been shown to have higher plant and animal biodiversity than adjacent national parks, where lightning-caused fires are allowed to burn unchecked but where aboriginal burning is prohibited (Yibarbuk et al. 2001, Fraser et al. 2003).

So if we conservatively assume that each Native American purposefully set only 1 fire/person per year, and that there were only 2 million native people north of Mexico, the aboriginal burning rate would have been 2.7–350 times greater than known lightning ignition rates (Table 1). If 10 fires/person per year were set, possibly a more realistic assumption (Boyd, T., 1986; Turner 1991; Gottesfeld 1994; Boyd, R., 1999; Anderson 2005), the aboriginal burning rate would have been 27–3,500 times greater than known lightning ignition rates. If there were 20 million Native Americans, instead of 2 million, that would add another order of magnitude to the estimated rate of purposefully set fires. Finally, if estimates of accidentally started aboriginal fires are combined with estimates of purposefully set management fires, the overall aboriginal burning rate would have been 2–5 orders of magnitude greater than known lightning ignition rates. Even if we assume there were no more than 500,000 native people in the United States and Canada, aborig-

inal ignition rates would still have overshadowed lightning fires. Thus, there have been more than enough people in the Americas for the past 10,000 or so years to completely alter fire regimes and vegetation patterns.

Moreover, widespread aboriginal burning, by consuming fuels and creating patches of burned and unburned vegetation, limited the spread and extent of any lightning fires that may have started, similar to what has been documented in Australia (Kay 1998, 2000). This would suggest that lightning-caused fires have been largely irrelevant in structuring plant communities throughout many areas in North America. It also turns out that it does not require very many native people to completely alter fire regimes because lightning ignition rates were so low and aboriginal ignition rates so high.

EXTENT OF ABORIGINAL BURNING AND VEGETATION MODIFICATION

There are several ecological examples that suggest aboriginal burning not only structured a wide range of plant communities but actually created many of the vegetation associations heretofore thought to be “natural.” Perhaps the most compelling evidence is from eastern United States forests.

For the last 8,000–10,000 y, much of the east-central United States was dominated by oaks (*Quercus* spp.), American chestnut (*Castanea dentata*), and pines (*Pinus* spp.), all fire-tolerant, early to mid-successional species (Delcourt et al. 1986, 1998; Clark and Royall 1995; Cowell 1995, 1998; Olson 1996; Delcourt and Delcourt 1997, 1998; Bonnicksen 2000). Since European settlement, however, oaks and pines have increasingly been replaced by late-successional, fire-sensitive species, such as maples (*Acer* spp.), even in protected areas (Botkin 1990:51–71; Abrams 1998, 2003, 2005; Batch et al. 1999; Bonnicksen 2000; Rodewald 2003; Roovers and Shifley 2003; Aldrich et al. 2005; Rentch and Hicks 2005). This and related fire-history studies suggest that the species composition of eastern forests had been maintained for thousands of years by frequent landscape-level burning (Black et al. 2006, Stambaugh and Guyette 2006). Now, this portion of the United States does have one of the highest lightning-strike densities in North America (Orville and Huffines 2001, Orville et al. 2002) but as noted in Table 1, these forests have the lowest lightning-fire ignition rates in the country. This is because when most lightning strikes occur during June, July, and August, eastern deciduous forests are often too green or wet to burn. In fact, eastern deciduous forests will readily burn only when the trees are leafless and the understory is dry—conditions that occur late in the fall, during winter, or early in the spring; all times when there are virtually no lightning strikes and hence no lightning-caused fires.

Thus, the only way for eastern forests to have displayed the open stand characteristics and species composition that were common at European settlement is

if those communities had regularly been burned by native people as part of aboriginal land management activities (Kay 2000, Mann 2005). Without humans actively managing these systems, the forests would be entirely different. It is also likely that aboriginal burning created the many eastern prairies and “barrens” reported by early Europeans (Campbell et al. 1991, Belue 1996, Barden 1997, Bonnicksen 2000, Mann 2005). Canebrakes (*Arundinaria gigantea*), too, likely owed their existence to native burning and other aboriginal land management practices (Platt and Brantley 1997).

Southern Canadian Rockies

Fire-history studies and repeat photographs both indicate that Banff and Jasper national parks once experienced a high frequency of low-intensity fires. Since the parks were established, however, lightning-caused fires have been exceedingly rare. In some vegetation types, fire return intervals are now 100 times greater than they were in the past (Wierzchowski et al. 2002). Lower montane valleys that once burned every 5 y or less now do not burn at all. Based on this and other evidence, Parks Canada has concluded that native burning, not lightning-caused fires, was critical in maintaining what heretofore was believed to be the “natural” vegetation mosaic of the southern Canadian Rockies (Kay et al. 1999). That is to say, there simply are not enough lightning-caused fires to account for historical burn and vegetation patterns (Wierzchowski et al. 2002).

Yellowstone National Park

Prior to park establishment, Yellowstone’s northern grasslands had a fire return interval of once every 25 y (Houston 1973). Yellowstone has had a “let burn” policy for over 30 y now, yet during that period, lightning-caused fires have burned practically none of the northern range. In 1988, fire did burn approximately one-third of the area, but according to agency definitions that was “unnatural” because the fire was started by man, not by lightning. Besides, the 1988 fires are thought to be a 100- to 300-y event, so similar fires could not have caused the original 25-y fire frequency (Kay 2000). Lightning strikes occur frequently on the northern range, but when they do during June, July, and August, the herbaceous vegetation is usually too green to carry a fire (Kay 1990). Thus, it is likely that the park’s original 25-y fire frequency was entirely the product of aboriginal burning.

Aspen Ecology

Repeat photographs and fire-history studies indicate that western aspen (*Populus tremuloides*) communities burned frequently in the past, yet experience has proven that aspen is extremely difficult to burn (Brown and Simmerman 1986). Terms such as “asbestos type” and “firebreak” are often used to describe aspen (DeByle et al. 1987:75). Even raging crown fires in coniferous forests seldom burn adjacent

aspen communities (Fechner and Barrows 1976). At current rates of burning, “it would require about 12,000 years to burn the entire aspen type in the West” (DeByle et al. 1987:73). Something is clearly different today from what it was in the past.

Research has shown that aspen communities will readily burn only when the trees are leafless and understory plants are dry, conditions that occur only during early spring and late in the fall (Brown and Simmerman 1986). Prior to 15 May and after 1 October, though, there are few lightning strikes and virtually no lightning fires in the northern or southern Rocky Mountains (Kay 1997, 2000, 2003). So, if aspen burned at frequent intervals in the past, as fire-frequency data and historical photographs indicate it did, then the only logical conclusion is that those fires had to have been set by Native Americans.

San Juan Mountains

Researchers in the southern Rockies contend that fire-history data obtained from fire-scarred conifers do not support the idea that aboriginal burning had any significant influence on “natural” fire regimes (Allen 2002, Vale 2002). Grissino-Mayer et al. (2004:1708), for instance, reported that they could find “no compelling evidence that Native Americans influenced fire regimes” in Colorado’s San Juan Mountains. Lightning-fire data, though, do not support that conclusion. According to Grissino-Mayer et al. (2004:1716), “57% of all fires prior to 1880 occurred during the spring dormant season” based on microscopic analysis of when fire scars were actually formed. Yet lightning-fire occurrence data provided by the National Inter-agency Fire Center show that only 11% of lightning fires occur during that period, and they account for only 3% of the area burned (Figure 3). This would suggest that something other than lightning was responsible for the earlier fires (Kay 2000:20–21).

Northern Great Plains

Baker (2002:51–66) questioned the validity of using early historical accounts to support the idea that native people routinely used fire to manage their environment. According to Baker, few Europeans actually observed Native Americans setting the fires that early explorers attributed to native people and early explorers were also ignorant of the role lightning played in starting fires, when they attributed fire after fire to aboriginal ignitions. In addition, Baker (2002:52) claimed that Europeans were biased in attributing fires to natives because whites wanted “to paint . . . Indians as reckless savages and poor land stewards who did not deserve to keep their land.” That is to say, because Europeans thought fires were “bad,” attributing landscape burning to native people would put aboriginal inhabitants in an unfavorable light. While there is some truth in this argument (Decker 2004), alternatively, early explorers could have attributed most fires to native people because native people started most fires (Pyne 2003).

One way to answer the questions raised by Baker

is to look at the current distribution of lightning-caused fires and to compare those data with observations from the early 1800s. Higgins (1984) reported that the majority of lightning fires on the northern Great Plains occur during June, July, and August (Figure 4a). Currently, there are few lightning-caused fires early in the spring or late in the fall because there are few lightning strikes outside of June, July, and August. Alexander Henry the Younger (Gough 1988) manned a trading post on the northern Great Plains from 1800 to 1807, and in his daily journal he recorded when the surrounding prairies were on fire. Henry observed prairie fires early in the spring and late in the fall but failed to report a single fire during June, July, or August (Figure 4b).

Vegetation on the northern Great Plains is often too green to burn during the June, July, and August growing season, but during droughts, lightning can set the prairies on fire during those months—these are the fires we see today. In the past, though, fire commonly swept the northern plains during early spring and late fall when the grasses are normally cured-out. Because there are virtually no lightning strikes early in the spring or late in the fall, all the fires reported by Alexander Henry the Younger likely were set by native people, whether Henry actually saw natives set those fires or not.

Then, too, there is Peter Fidler’s journal (Haig 1992), a source not cited by Baker (2002). During the winter of 1792–1793, Peter Fidler traveled with a band of Piegan natives from Buckingham House east of present-day Edmonton, Alberta, to the Oldman River just north of the U.S. border. Fidler entered the southern Canadian Rockies and his journal is the earliest, firsthand, European description of the Rocky Mountains. Fidler repeatedly described how native people, both inadvertently and purposefully, set the plains on fire. And, most amazingly, during winter, well outside what is today the “normal” burning season. As there are no lightning strikes on the northern Great Plains during winter, every fire reported by Fidler must have been set by native people.

In addition, Fidler reported that the plains were commonly afire during spring and fall, but he made a mistake by attributing the spring and fall fires, which he did not personally observe, to lightning, and not to natives (Haig 1992:36). As there are no lightning fires on the northern Great Plains during spring or fall (Higgins 1984, Wierzchowski et al. 2002), all the burning reported by Fidler can be attributed to native people. In the spring of 1793, Fidler left the southern Alberta prairies and returned to Buckingham House, a journey of approximately 480 km. Over that distance, Fidler reported that they could find virtually no unburned ground on which to pasture their horses, such was the extent of aboriginal burning:

Grass all lately burnt the way we have passed this Day towards the Mountain, but not to the South of us, but at a good distance in that direction the Grass is now burning very great fury, supposed to be set on fire by the Cotton

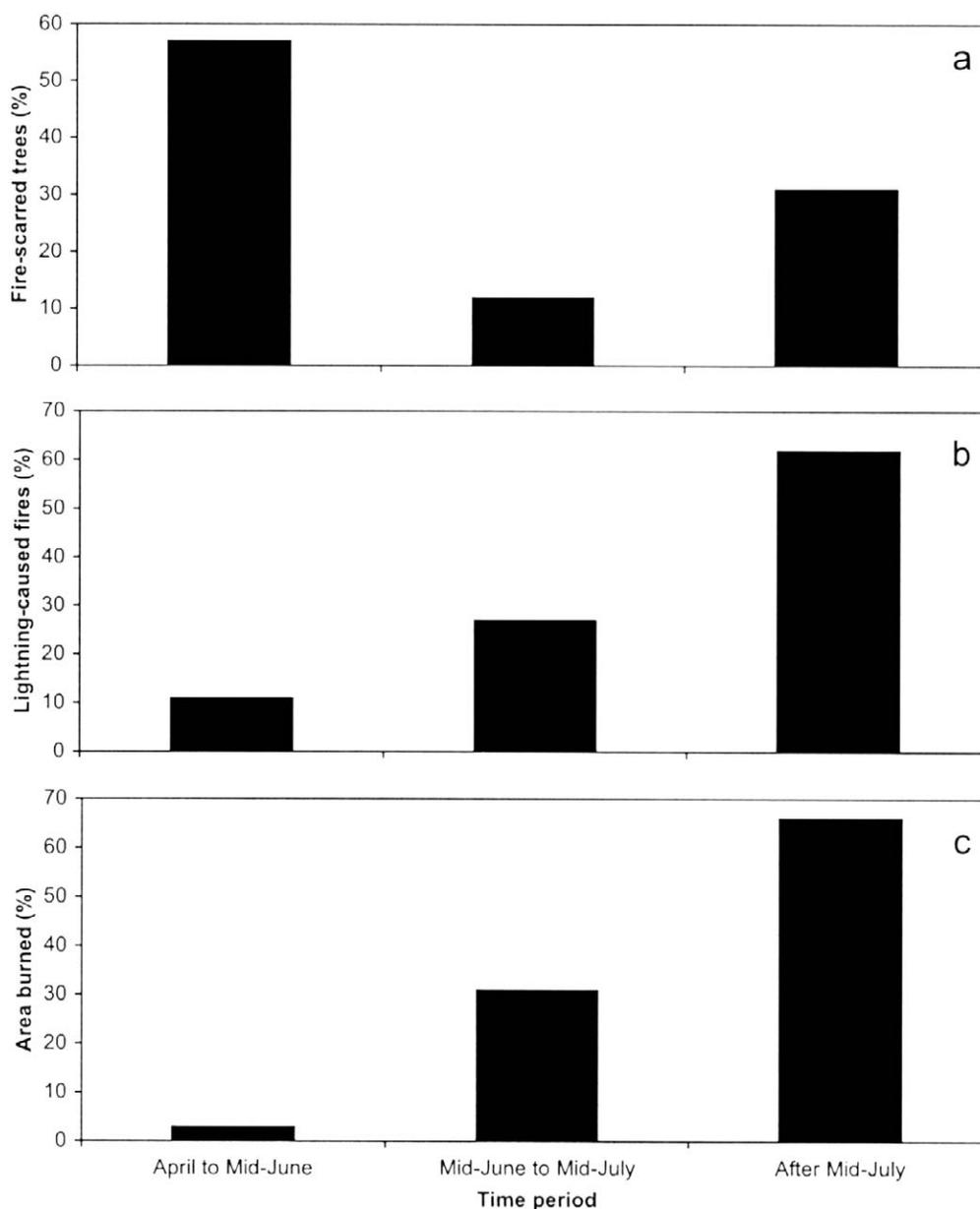


Fig. 3. Fire history of Colorado's San Juan Mountains. (a) The proportion of trees scarred by fire during different time periods prior to 1880, as reported by Grissino-Mayer et al. (2004). (b) The distribution of lightning-caused fires reported on the San Juan National Forest from 1970 to 2002. Data provided by the National Interagency Fire Center, Boise, ID. (c) The distribution of the area burned by lightning-started fires on the San Juan National Forest from 1970 to 2002. Data provided by the National Interagency Fire Center, Boise, ID.

na hew Indians. Every fall & spring, & even in the winter when there is no snow, these large plains either in one place or other is constantly on fire, & when the Grass happens to be long & the wind high, the sight is grand & awful, & it drives along with amazing swiftness [Haig 1992:36].

West Coast Forests

Frequent fires once shaped many coastal forests in northern California, Oregon, and Washington. Coastal redwoods (*Sequoia sempervirens*), for instance, historically were visited by fire every 10–20 y or less

(Brown and Baxter 2003, Stephens and Fry 2005). Frequent fire also once maintained a multitude of prairies, balds, and open areas within the forest mosaic (Zybach 2003). Lightning fires in these forests, however, are virtually nonexistent and these areas have some of the lowest lightning-fire ignition rates in the West (Table 1). Thus, many ecologists and anthropologists attribute the earlier burning to native people, who used fire to improve the productivity of various plant communities (Norton 1979, Boyd 1986, Lewis 1990, Liberman 1990, Brown and Baxter 2003, Wray and Anderson 2003, Zybach 2003, Anderson 2005, Carloni 2005, Keeley 2005, Stephens and Fry 2005). In the absence of regular native burning, prairies and

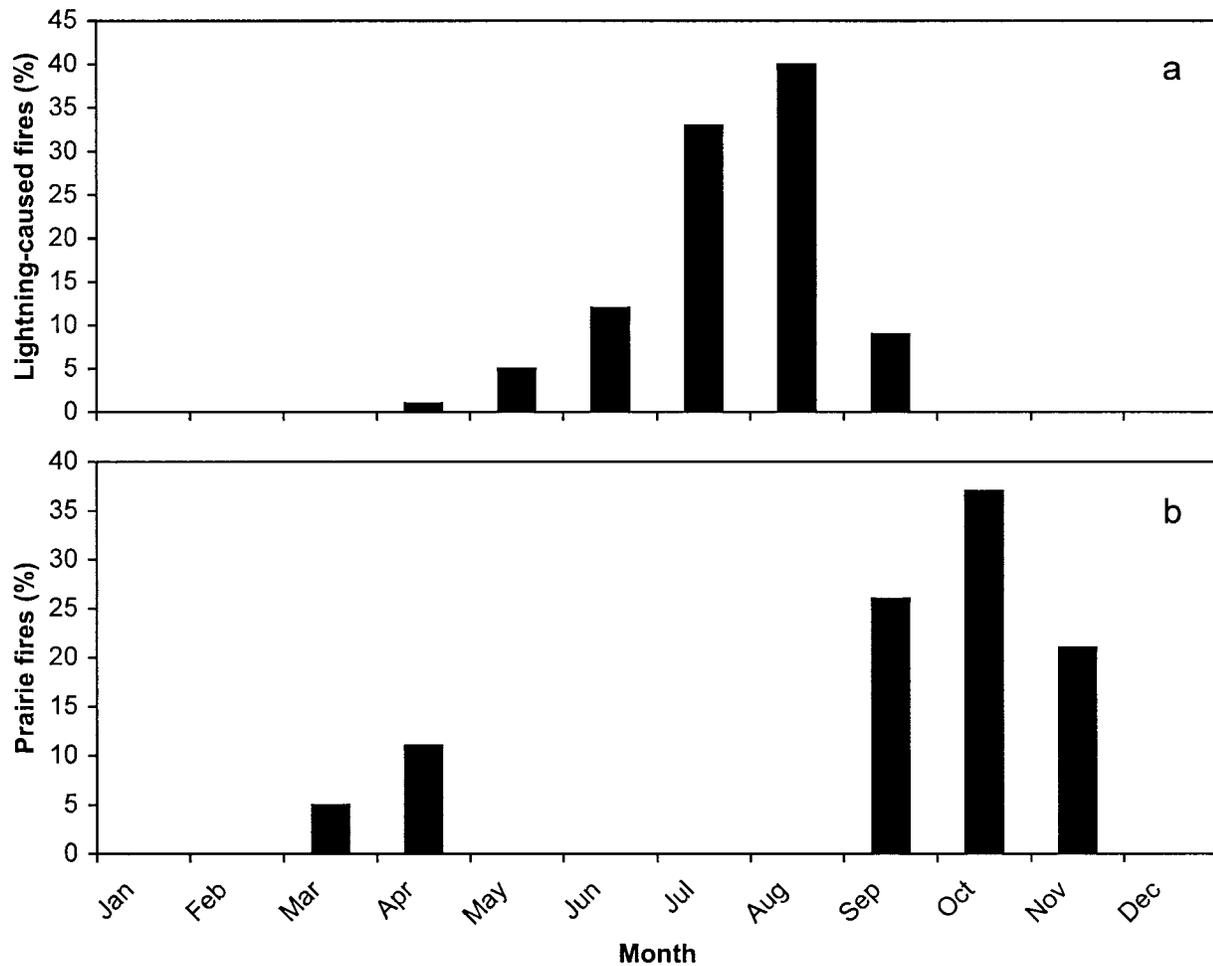


Fig. 4. The distribution of fires on the northern Great Plains. (a) The distribution of lightning fires that occurred from 1940 to 1981, as reported by Higgins (1984). There are few lightning fires during spring or fall because there are very few lightning strikes during those periods. (b) The distribution of prairie fires as reported by Alexander Henry the Younger from 1800 to 1807 when the northern Great Plains were under aboriginal control (Gough 1988).

balds are now overrun by encroaching conifers. The entire Willamette Valley, for instance, which was largely a grassland at European contact, reverts to forest in the absence of regular burning (Habeck 1961, Boyd 1986, Zybach 2003).

Whitlock and Knox (2002), though, contend that declining fire frequencies are due to climatic change and that, historically, aboriginal burning was unimportant. Whitlock and Knox, however, failed to explain how global climatic circulation patterns could change to such an extent that lightning-strike densities would increase in coastal areas. Moreover, even if known lightning-fire ignition rates were 100 times higher in the past, they would still have been overshadowed by human ignition rates, as coastal areas of northern California, Oregon, and Washington were densely populated by a vast array of aboriginal people due to abundant stocks of salmon (*Oncorhynchus* spp.), vegetal foods, and marine resources (Zybach 2003). Whatever climatic changes may have occurred were inconsequential given the level of aboriginal burning that existed.

First Contact

A similar debate has been going on for many years over what caused Pleistocene megafaunal extinctions as modern humans spread out of Africa (Kay and Simmons 2002). One school holds that climatic change drove the extinctions, while the other contends that humans killed-out the megafauna in the Americas and around the world—see Kay (2002) for a detailed discussion of this debate.

To separate between these competing hypotheses, Miller et al. (2005) looked at carbon isotopes in emu (*Dromaius novaehollandiae*) eggshells and wombat (*Vombatus* spp.) teeth—records that span 150,000 y in Australia. Miller et al. (2005) reported an abrupt change in feeding habitats 45,000–50,000 y ago when humans first colonized Australia. As noted by Johnson (2005:256), “The fact that the distributions and feeding habits of both species changed so little in response to climate extremes, but so much when people arrived, tells us that the impact of human arrival far exceeded the effects of any of the climate changes of the past 140,000 years.” Miller et al. (2005:290) suggested,

“that systematic burning practiced by the earliest human colonizers may have converted a drought-adapted mosaic of trees and shrubs intermixed with palatable nutrient-rich grasslands to the modern fire-adapted grasslands and chenopod/desert scrub.” Similarly, Robinson et al. (2005:295) reported a sharp rise in charcoal recovered from sediment cores at the time humans initially colonized eastern North America and suggested that this represented anthropogenically driven “landscape transformation” on a grand scale. As humans drove the megafauna to extinction by hunting, escaped campfires and purposeful burning completely reconstituted vegetation communities.

CONCLUSIONS

According to Parker (2002:260), who discounted the ecological impact of aboriginal burning, “nostalgia and political agendas are no substitute for valid evidence,” and I concur, as do others (LaLande 2003, Pyne 2003). The evidence suggests that lightning-caused fires were never more frequent than native-set fires—either escaped campfires or purposefully started fires at even the lowest aboriginal population estimates. Various ecological examples also suggest that native burning was a much more important ecological factor than lightning-caused fires. There is also the problem that reported fire return intervals do not present a true measure of how often areas once burned. It has been known for some time that low-intensity surface fires, which were the norm in many ecosystems prior to European settlement, do not scar each tree they burn, even if that tree had been previously scarred.

The only experimental data that I have been able to locate are for oaks in eastern forests where researchers repeatedly prescribed-burned stands at 1-, 2-, or 3-y intervals and then cut down the trees to count fire scars (Smith and Sutherland 1999, Sutherland and Smith 2000). On average, only one-third of burned trees were actually scarred by fire (Elaine Sutherland, U.S. Forest Service, personal communication). Similarly, Skinner and Taylor (2006) noted that 86% of Douglas-fir (*Pseudotsuga menziesii*) stumps with internal fire scars had no external evidence of the trees having been burned. When those hidden fire scars were taken into account, the estimated fire return interval declined by nearly 50% (Skinner and Taylor 2006:204–206), while Shirakura (2006) observed that only one in seven fires were recorded by oaks in east-central Oklahoma. This would suggest that published fire-history studies tend to underestimate the true frequency of burning.

How often did areas burn in the past? As often as native people wanted them to burn. There is little doubt that Native Americans fully understood the benefits they could receive by firing their environment (Anderson 2005). To suggest otherwise is to assume aboriginal people were ecologically incompetent (Andersen 2005), a supposition that is not supported by any reading of the historical or ethnographic record

(Mann 2005). Thus, the idea that the Americas were a pristine wilderness, untouched by the hand of man (Vale 2002) is a statement of belief, not a fact supported by science (Kay 2002, Pyne 2003).

Finally, this paper is a first attempt at estimating how many fires native people may have started and, as such, I did not consider cultural differences or how aboriginal burning may have varied over time, under different subsistence strategies, or by area. I also assumed that native people were systematically distributed across the landscape, which was surely not the case with more settled societies. Nevertheless, even with the simplifying assumptions that were employed, aboriginal use of fire most likely overwhelmed lightning ignitions as Stewart (1956, 1963, 2002), Anderson (2005), and others contend.

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NATIVE AMERICAN FIRE PATTERNS IN YOSEMITE VALLEY: A CROSS-DISCIPLINARY STUDY

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ABSTRACT

The inability to distinguish between human-caused and lightning ignitions in fire-history studies has led to three major problems: 1) a basic assumption that all pre-Euro-American settlement fire regimes are “natural” unless findings are aberrant, i.e., outside the range of “natural” lightning fire regimes; 2) a lack of studies that explicitly or quantitatively determine ignition sources; and 3) use of regional anthropological overviews rather than site-specific ethnographic and archaeological data.

A cross-disciplinary dendrochronological fire history and archaeological study conducted in Yosemite Valley, Yosemite National Park, California, shows that fire return intervals in areas with no historical lightning ignitions and a large Native American population were similar to those in locations with a high number of lightning ignitions. Native American fire regimes in Yosemite Valley consisted of spatially small, low-intensity surface fires in all areas regardless of differences in distance from a village site, identified land uses, or village size. Fire patterns appear to be independent of climatic fluctuations and dependent on human disturbance patterns. Archaeological and ethnographic data show no major difference between the population size, land-use patterns, or material culture of the Ahwah’-nee, the prehistoric occupants of Yosemite Valley, and other native groups in the Sierra Nevada or Great Basin. The cultural data and initial findings from this study suggest that lightning and Native American influences on fire regimes cannot be differentiated based only on fire return intervals and fire regimes; additional cross-disciplinary studies are needed to gain better understanding of human–fire interactions.

keywords: anthropogenic fire regimes, archaeology, California, fire history, mixed conifer, Sierra Nevada, Yosemite Valley.

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INTRODUCTION

Within California and the Sierra Nevada, anthropological studies have shown that Native Californians used fire to manipulate vegetation for various reasons (Reynolds 1959, Lewis 1973, Pyne 1983, Gruell 1985, Roper Wickstrom 1987, Anderson 1988, Anderson and Moratto 1996, Williams 2003). Ethnographic data from throughout North America illustrate 11 broad cross-cultural categories or motives for fire use: hunting, crop management to increase growth and yields, fire proofing areas, insect collection, pest management, warfare or economic extortion, clearing areas for travel, felling trees, clearing riparian areas, “careless campfires,” and ritual use (Stewart 1956, Lewis 1973, Barrett 1981, Anderson 1988, Bonnicksen et al. 1999, Williams 2003).

The debate over Native American use of fire and its effects is not new, but over recent years has re-emerged as a research topic. It is often a polarizing debate, and as an attempt to bring greater understanding to this topic, this study used both a dendrochronological fire history and anthropological studies to test common assumptions from both sides of the debate. The first step is to test the basic assumption of whether we can distinguish between a “natural” light-

ning fire regime and an anthropogenic fire regime through fire history alone.

From the viewpoint of an anthropologist, fire histories assume all pre-Euro-American settlement fire regimes are “natural” unless findings are aberrant, i.e., outside the range of lightning fire regimes. It is often assumed that anthropogenic fire signatures are outside of the natural fire regime. For the purposes of this paper, *natural fire regime* is that obtained through lightning ignitions only. Often it is implied that anthropogenic fire regimes will be different due to either shorter or longer fire return intervals than produced by lightning alone, fires occurring during seasons when lightning does not generally occur, and fires that are asynchronous with climate trends (i.e., fire in wet years and no fire in dry years).

These ideas have not been strenuously tested because the majority of published dendrochronological fire history studies have not explicitly or quantitatively determined ignition sources; and such studies use broad regional anthropological overviews rather than site-specific ethnographic and archaeological data. In this study, I test those assumptions using fire-scar data and archaeological data.

STUDY AREA

As a first step to understanding how one Native American tribal group may have burned the landscape, I sought a location where fire scars could be attributed

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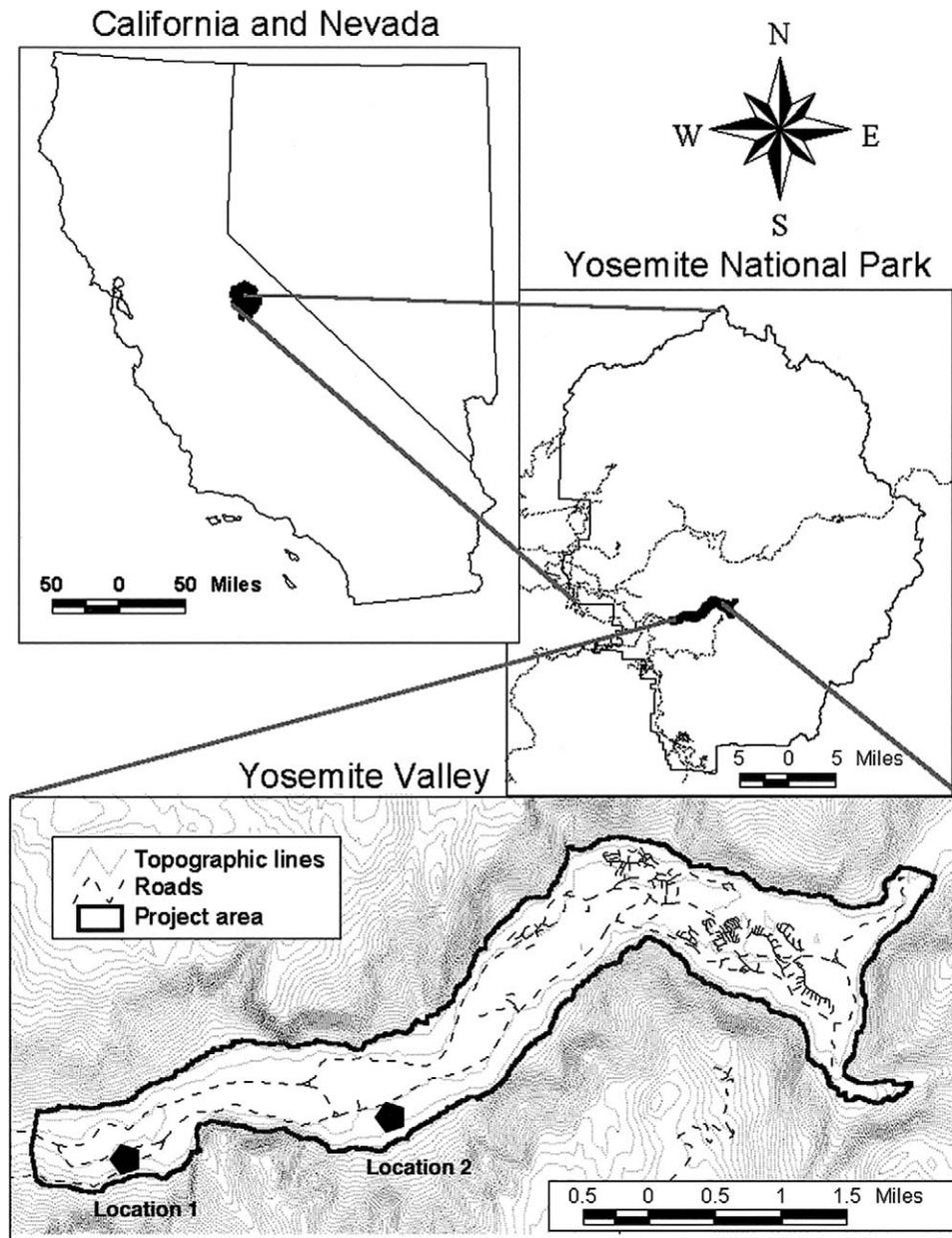


Fig. 1. Location of study sites in Yosemite Valley, Yosemite National Park, California.

largely to ignitions by humans, due to a lack of lightning ignitions in the area, and where ethnographic and archaeological data were available to determine the spatial relationship of fire to known periods of occupation. Such a location would have a unique geography that lacked lightning-ignited fires, ethnographic data with known village sites and gathering areas, and archeological data that supported the ethnographic data.

Yosemite Valley (37.7438°N , 119.5900°W [NAD83/WGS84]), in Yosemite National Park, on the western slope of the central Sierra Nevada of California, fit these criteria. Located approximately 250 km east of San Francisco, Yosemite Valley was a glacially carved valley approximately 1.6 km wide by 11.4 km long (Figure 1). The valley was a U-shaped cross section, with broad expanses of glacially scoured granite

cliffs topped by exposed bedrock domes. The valley floor was at approximately 1,200 m in elevation.

The climate of the western slope of the Sierra Nevada was montane, with cool, moist winters and warm, dry summers. The high elevation of the Sierra Nevada presented a barrier to the pattern of prevailing storms moving east from the Pacific Ocean. Winter snows occurred in areas above approximately 1,065 m; Yosemite Valley was often snow-free, allowing year-round occupation (Hull and Kelly 1995). Yosemite Valley was located within the Sierran Montane Forest Mixed-Conifer vegetation type.

Ethnographic studies have shown that the Southern Sierra Miwok, who inhabited Yosemite Valley, used fire for crop management for at least 250 different plants (Barrett and Gifford 1933; Anderson 1988, 1993; Bibby 1994). Ethnographic work conducted

within Yosemite Valley identified 23 plants obtained from 35 traditional gathering locations (Bibby 1994).

While the Southern Sierra Miwok probably used fire for all 11 of the cross-cultural motives previously listed, the topographic limitations of Yosemite Valley and cultural needs of the Ahwah'-nee probably made crop management the most common reason for using fire in this area. Fire was one tool used as part of intensive individual plant- or patch-level management practices that were required to produce the highest quantity of materials required for subsistence, basketry, cordage, and building (Anderson 1988).

Lightning Ignitions in Yosemite Valley

Since records have been kept, beginning in 1930 through 2002, no lightning-ignited fires were recorded on the floor of Yosemite Valley (National Park Service 2002). Modern fire-history data indicate that fewer lightning-caused fires occurred in the valley floor than in other areas of similar elevation within the park. Yosemite National Park contained approximately 256,975 vegetated burnable ha (635,000 acres) and averaged 41.5 lightning fires per year from an average of approximately 1,200 lightning strikes (van Wagtenonk 1993). Between 1930 and 2002, 2,877 lightning fires burned 69,759 ha (172,379 acres) within Yosemite National Park (National Park Service 2002). Yosemite Valley was located within the elevational range of 910–1,830 m, which had a high density of lightning-strike fires (van Wagtenonk 1993). A geographic information system (GIS) overlay of modern fire-history data (National Park Service 2002) and the study area showed that no lightning-ignited fires started or spread into the study area between 1930 and 2002.

While no published studies have reported the lack of lightning-ignited fires within Yosemite Valley, absence of such fires, in part, was likely due to the topography of the valley and its surrounding cliffs. Lightning strikes are a function of topography and gravitate toward high points, with ridge tops and mid-slope prominences being the most likely struck, features that are decidedly lacking on a valley floor (Kornarek 1967). In a study from the Yosemite region, Reynolds (1959) found that of 319 lightning fires, 83% occurred in either the top or middle portion of a slope. The high granite walls that surrounded Yosemite Valley probably received the vast majority of lightning strikes in the valley area, resulting in more fires on the valley rim, above the valley floor, and outside the study area. Why the relatively few lightning strikes that occurred on the floor of Yosemite Valley did not cause fires at the same rate as in other areas is beyond the scope of this study.

Ethnographic and Archaeological Overview

Ethnographic studies show that Yosemite Valley was within the traditional territory of the Southern Sierra Miwok, although several other groups traveled to the valley for purposes of trade and possibly for limited periods of residence (Bennyhoff 1956, Hull and Kelly 1995). The Sierra Miwok were divided into

Table 1. Proposed cultural chronology for Yosemite Valley, California (after Moratto 1999).

Date	Period	Phase
Post AD 1945	Historic 4	
AD 1891–1944	Historic 3	
AD 1864–1890	Historic 2	Rancheria
AD 1848–1863	Historic 1	Tenaya
AD 1800–1847	Protohistoric	Yosemite
AD 1350–1800	Late Prehistoric 3	Mariposa
AD 650–1350	Late Prehistoric 2	Tamarack
1200 BC–AD 650	Late Prehistoric 1	Crane Flat
3500–1200 BC	Intermediate Prehistoric 2	Merced Wawona
6000–3500 BC	Intermediate Prehistoric 1	Unnamed
7500–6000 BC	Early Prehistoric 4	El Portal

tribelets. A tribelet consisted of approximately 100–300 people. Each tribelet controlled the natural resources within a defined territory and inhabited several permanent settlements and a larger number of seasonal campsites (Gifford 1916, Broadbent 1960). Merriam identified the inhabitants of Yosemite Valley as “the Ahwaneech or Ahwah'-nee Mew'-wah” (Merriam 1917). According to Merriam (1917), occupation in Yosemite Valley consisted of both permanent villages occupied throughout the year and summer camps used from May through October, and the valley was “somewhat depleted in the winter” when the residents moved down to the Merced River Canyon.

Numerous archaeological investigations have identified 130 sites of which >100 contain evidence of Native American use (Bennyhoff 1953, 1956; Groscup 1954; Napton et al. 1974; Napton 1978; Hull and Kelly 1995). The archaeological evidence suggests that areas within Yosemite National Park have been inhabited since 7500–6000 BC. Currently, the first occupation observed within Yosemite Valley was radiocarbon-dated to 5200 BP (Moratto 1999). Archaeological data show that Yosemite Valley has been occupied continuously since this period. Changes in material culture suggest that the Miwok arrived in Yosemite Valley at approximately 700–450 BP (Moratto 1999). The material culture from the Yosemite region has suggested nine major periods or phases of occupation (Table 1).

METHODS

Study Site Selection

Study sites consisted of a village paired with a gathering area and a fire-history “control” area in close proximity to each other. Within each study site, village, gathering, and control areas had no major topographic or geological barriers to ground fire spread (e.g., the gathering area could not be on a forested island within a landslide or river, separated from its paired village and control). These pairings capture the gradient of human land use from human occupation to resource extraction to nonuse.

Villages had to correspond with a village site identified in historic or ethnographic literature or field notes, archaeological data had to place occupation of

the site partially or entirely within the latter portion of the Late Prehistoric 3, Protohistoric, or Historic 1–3 periods. Traditional gathering areas must have been identified in Bibby (1994) and controls must have lacked evidence of archaeological occupation and lacked evidence of use as a traditional gathering area. All areas were required to have 10–30 fire-scarred trees in or immediately surrounding the sampling area.

Historic and ethnographic records, including the work of Stephen Powers (1871–1876), S.A. Barrett and E.W. Gifford (1906–1920s), C. Hart Merriam (1900–1920s), Brian Bibby (1994), and M. Kat Anderson (1980s–1990s), were reviewed for information on village locations and land-use patterns (Gassaway 2004). All location-specific information was transferred into GIS. Historic and ethnographic maps were scanned and georeferenced using ArcView 3.2a and Arc/Info 8.02 (ESRI, Redlands, CA).

Archaeological surveys and excavation data were overlaid with the historic and ethnographic data layers to determine the known physical manifestation of Native American occupation and land use.

Based on the spatial analysis of human occupation, I conducted fire-history surveys focused on areas in and around each village to locate well-preserved, fire-scarred trees. Fire-scarred trees were mapped using a Global Positioning System (GPS) unit (Garmin, Olathe, KS; Trimble, Sunnyvale, CA; Magellan, Santa Clara, CA). GPS point locations were converted into a density grid of fire-scarred trees per hectare. The density grid was overlaid with ethnographic village sites, gathering areas, and prescribed fire locations (1970–2002).

Based on potential for numerous fire scars and spatial correspondence with ethnographic, archaeological, and gathering areas, three study sites were chosen for fire-history sampling and further archaeological investigations.

Archaeological Investigations

Surface archaeological constituents at the three study sites were reevaluated to determine if archaeological excavations were needed to refine period of occupation and extent of use. At one location, the extensive surface artifacts confirmed that occupation occurred throughout the Late Prehistoric, Protohistoric, and Historic periods. Two locations had minimal surface artifacts that were unreliable for dating, and limited archaeological testing took place to better define periods of occupation. Archaeological testing consisted of surface scrapes and shovel tests placed on a north–south and east–west grid that bisected at 5- or 10-m intervals. Surface scrapes consisted of 1 × 1-m units where all duff was removed to expose mineral soil. Shovel tests were 50 × 50 cm and limited to maximum depth of 50 cm. All soils were screened using 1.6-mm (1/16-inch) screen. I fully analyzed and categorized all artifacts collected based on Wilson et al. (2002), except lithic materials, which were analyzed and categorized based on Byram (1996).

Fire History Methods

At each study site, 30–90 “cookies” and wedges from live and dead incense cedar (*Calocedrus decurrens*) and ponderosa pine (*Pinus ponderosa*) were collected. This consisted of 10–30 samples from each village, gathering area, and control. The Vegetation Dynamics Lab at Pennsylvania State University and I conducted analysis of the fire-scar samples. All samples were air-dried and sanded until ring structure detail was visible. Specimens were cross-dated by matching common patterns of wide and narrow rings in comparison to local master chronologies (King 1991, Stephens and Collins 2004). For cross-dating specimens, I used methods based on Stokes and Smiley (1996) and Pennsylvania State University used methods based on Yamaguchi (1991). Fire scars were identified using 1) the presence of a gap or break within a ring or along a ring boundary, 2) charred wood within the gap or break, and 3) subsequent overlapping curvilinear growth over the gap (Stokes 1980, Dieterich and Swetnam 1984, Savage and Swetnam 1990). Each fire scar was assigned a calendar year. Scar positions were assigned to one of six categories: 1) early earlywood (first one-third of earlywood), 2) middle earlywood (second one-third of earlywood), 3) late earlywood (last one-third of earlywood), 4) latewood (in latewood), 5) dormant (at ring boundary), or 6) undetermined (Baisan 1990, Kaye and Swetnam 1999).

Statistical differences in fire-history data between the sampling areas, villages, and time periods were determined with Student’s *t*-test, *F*-test, and two-sample Kolmogorov–Smirnov test produced using FHX2 software (Grissino-Mayer 2001). In order to determine when culture change or Euro-American influence may have affected Native American burning patterns, time periods developed by Moratto (1999) (Table 1) were used to compare changes of composite mean fire interval (MFI) over time. Statistical comparisons of each time period to subsequent periods were conducted using FHX2 software. To determine if patterns of anthropogenic fire use differed based on land-use patterns, fire-scar data were analyzed based on anthropological land-use types. At both study sites, fire-scar data were subdivided based their proximity to the village, gathering area, and control (Figures 2, 3).

RESULTS

Archaeological Excavations

Archaeological excavations were unable to locate any physical manifestations of Native American occupation at one study area. Surface manifestations and excavations found that two sites in the southwest portion of Yosemite Valley had Native American occupation during the Late Prehistoric 3, Protohistoric, and/or Historic 1–3 periods (Figure 1).

Location 1, the village of Sap-pah’sam-mah, was identified in the 1890s as “the lowermost (most westerly) village or camp on south side of the valley, about half a mile east of Pohono Meadows” (Merriam 1917: 205) (Figure 2). Archaeological evidence for Sap-

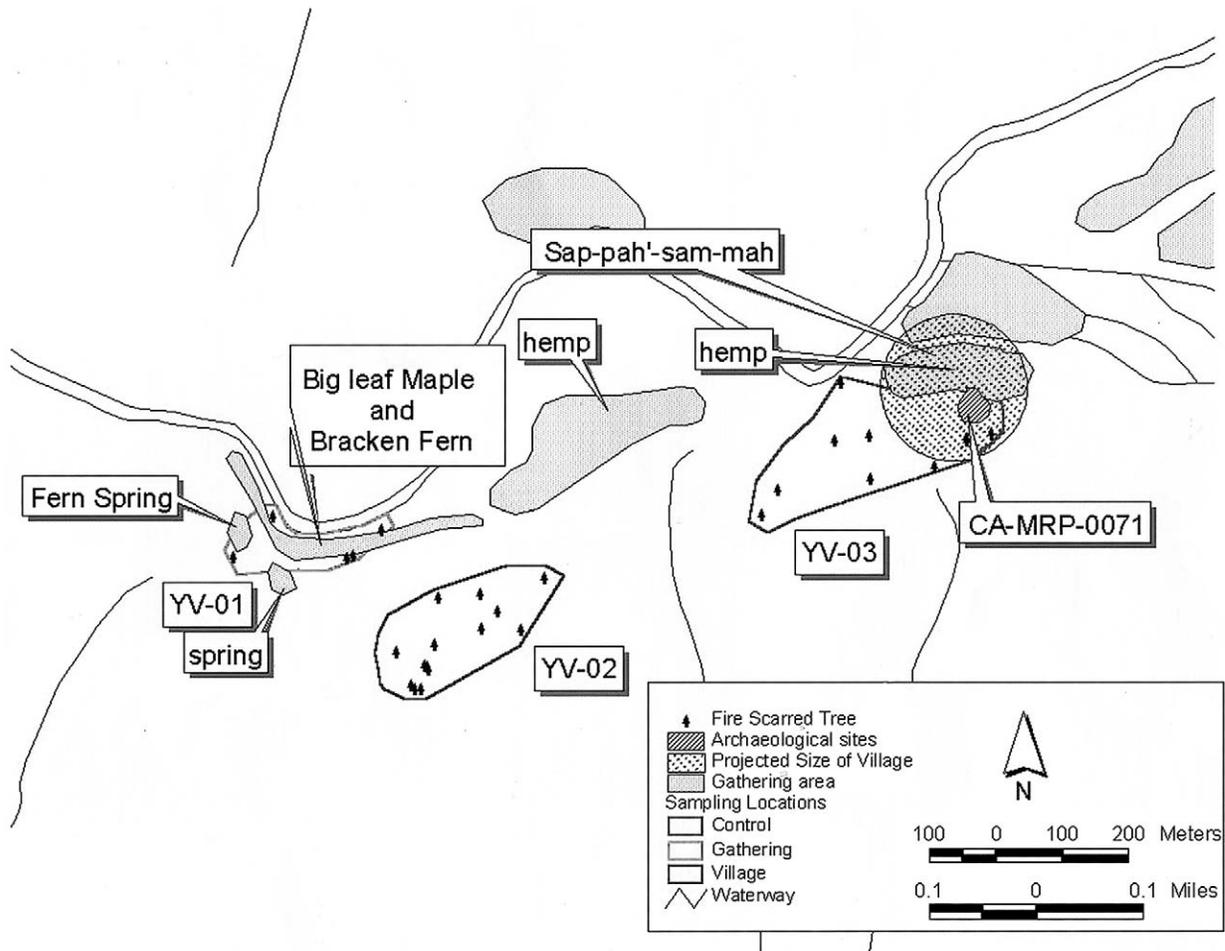


Fig. 2. Fire-scar samples from areas associated with the village of Sap-pah'-sam-mah (location 1), Yosemite Valley, Yosemite National Park, California.

pah'sam-mah consists of one archaeological site, CA-MRP-71. Artifacts from the site were consistent with two periods of occupation. A late prehistoric period consisting of small obsidian retouch flakes and debitage, one transverse side scraper, and one blue glass trade bead, were all consistent with Merriam's description of a seasonal camp with possibly seasonally low-intensity use and with a low diversity of use. A second period of use consisted of artifacts dating from ca. 1870 to 1960s, which is consistent with Euro-American occupation and tourism associated with the Wawona Road, which opened in 1875.

Location 2, the village of Kis'-se, is located mid-valley, south of the Merced River (Figure 3). The first documented use of this area by Native Americans occurred in 1879 on the Wheeler Survey map (Wheeler 1883, Hutchings 1886). The map indicates that the area northwest of Kis'-se and its adjacent gathering area was "Indian Pasture." The first written description of Kis'-se was by C. Hart Merriam (1917:207), who described Kis'-se or Kis's-se-uh it as a "large village near the river . . . Kis'-se was the westernmost of the large villages on the south side."

The village of Kis'-se consists of the archaeological site CA-MRP-76, which consists of midden soils,

two stationary milling outcrops with 50 mortar cups, and an obsidian debitage scatter (Hull and Kelly 1995). C. Hart Merriam's identification of Kis'-se, and surface constituents, are consistent with Protohistoric and Historic 1 occupation.

Fire History

The fire return intervals observed in the dendrochronological fire history, AD 1552–2004, revealed a composite MFI of 1.92 and a point MFI of 17.83 y with a range of 1–56 y (Figure 4; Tables 2, 3).

Significant differences in burning patterns occurred between the Late Prehistoric 3 (AD 1350–1800) and Protohistoric (AD 1800–1847) periods and the Historic 2 (AD 1864–1890) and Historic 3 periods (AD 1891–1944) (Table 4). The majority of land-use types showed no statistical difference between any of the other land-use areas by time period. The only statistical difference was between the bigleaf maple (*Acer macrophyllum*) and bracken fern (*Pteridium aquilinum*) gathering area, YV-01 (Table 3). When data from land-use types at both village sites were merged, no differences were detected. Each land-use type showed

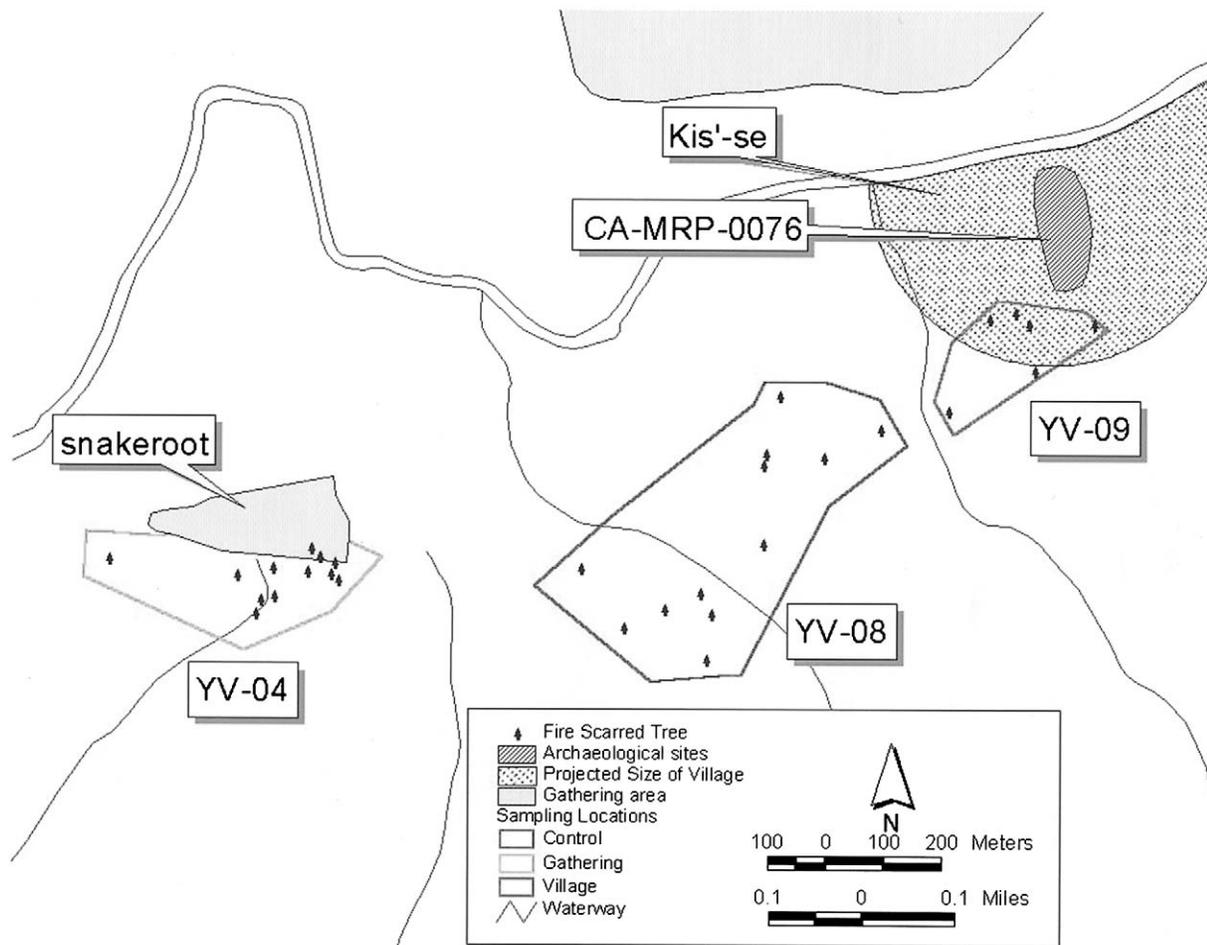


Fig. 3. Fire-scar samples from areas associated with the village of Kis'-se (location 2), Yosemite Valley, Yosemite National Park, California.

similar change in fire history over time, as did both study sites (Figure 5).

C. Hart Merriam (1917) suggested differences existed in the amount and extent of Native American use between the two study sites. Merriam identified occupation at location 1, Sap-pah'sam-mah, as a village or camp and location 2, Kis'-se, as a large village. The archaeological remains confirm a difference in the geographic extent of the two villages. Kis'-se (CA-MRP-76) encompassed 1.6 ha while Sap-pah'sam-mah (CA-MRP-71) extended only 0.2 ha. To determine if the village size had an effect on fire return intervals, all samples associated with Sap-pah'sam-mah (YV-01, -02, and -03) and Kis'-se (YV-04, -08 and -09) were combined and compared with a two-tailed *t*-test and a chi-square test.

The two-tailed *t*-test showed no significant differences in fire history during the time periods analyzed (Figure 5). The two locations were then compared for synchrony in fire dates to determine if fires observed at the two villages could have been produced by the same fire events. Only during five years (1775, 1783, 1800, 1841, 1864) did $\geq 10\%$ of the samples show potential for being produced by the same fire (Table 5). When the analysis was expanded to $\geq 25\%$ of the

samples scarred, no fire year showed the potential for being produced by the same fire.

The high frequency of fires and lack of synchronicity between villages suggests that the fires were spatially small, low-intensity surface fires and not crown fires. Visual observations of the spatial fire-scar data suggest that multiple small patches of 0.1–0.5 ha were burned each year.

DISCUSSION

Modern fire records kept between 1930 and 2002 show no lightning-ignited fires on the floor of Yosemite Valley, creating a modern lightning fire return interval of approximately 70+ y. Discussion of the cause of this lack of lightning-ignited fires on the floor of Yosemite Valley is beyond the scope of this paper. For this study, it is sufficient that there is a significant difference between the number of lightning-ignited fires and the fire return intervals in the dendrochronological fire history.

The fire regime surrounding Native American occupation was frequent, spatially small, low-intensity surface fires. These small fires were probably set in

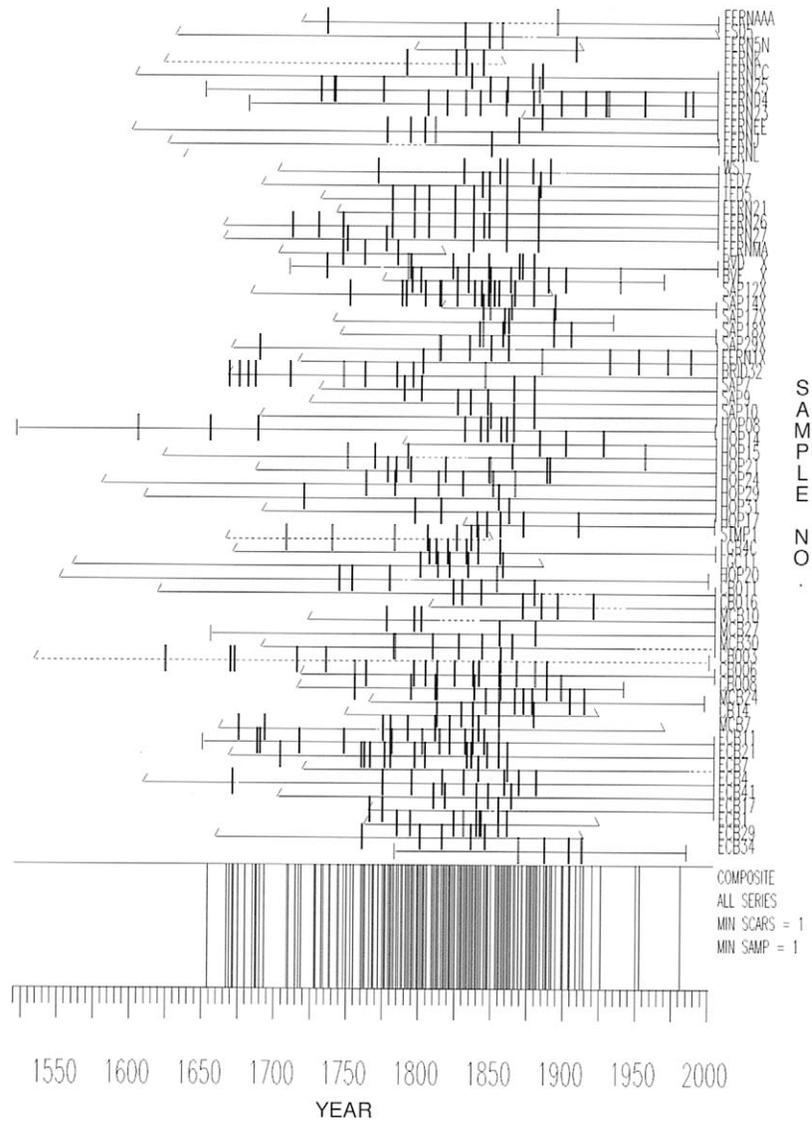


Fig. 4. Fire history for all samples, Yosemite Valley, Yosemite National Park, California.

rotation, creating a highly dynamic patchwork of different vegetation communities and communities in various stages of succession. This pattern was consistent regardless of land-use patterns. The one location that did show a difference, YV-01, had less to do with shifts in Native American fire use than a problem of sample size and proximity to a stable water source. Only five samples could be cross-dated from YV-01. The location was adjacent to the Merced River and a

spring, both stable water sources that created highly complacent tree rings. Complacent tree rings showed little variation based on climate and limited the ability to cross-date samples.

The fire frequency changed over time, and these changes were consistent with known cultural time periods and cultural changes observed in the archaeological record. Changes observed included longer fire intervals prior to AD 1800 and shorter fire intervals post-AD 1800. This suggests that, at least for the southwestern portion of Yosemite Valley, something impacted the Native American burning patterns during the late Prehistoric 3 period. The change was likely associated with the cultural change created by Spanish occupation of the California coast.

The decrease in fire return intervals shows that more fire was being applied to the southwestern portion of Yosemite Valley post-AD 1800. Following cultural changes that occurred prior to AD 1800, the use of fire remained stable throughout the early historic periods. This stability remained despite large-scale,

Table 2. Fire interval analysis by sample size, Yosemite Valley, California.

Variable	Study area	Sample area	Sample
Area (ha)	30	1.5–11	Tree
No. of samples	57	5–14	
Mean fire interval	1.92 ^a	4.69–17.83 ^a	17.7 ^b
Weibull median	1.61	3.33–16.5	
Interval range	1–11	1–43	2–56

^a Composite interval.

^b Point interval.

Table 3. Mean fire intervals based on land use during each archaeological time period, Yosemite Valley, California.

Land-use classification	All time periods	Time period				
		Late Prehistoric 3	Protohistoric	Historic 1	Historic 2	Historic 3
All samples	1.92	2.4	1.09	1.5	1.47	5
Gathering areas	4.62	5.13	2.35	2.2	5.75	
Controls	3.12	3.31	1.92	2	1.71	8
Village	3.9	3.96	1.62	1.75	3.29	9.75
Location 1	3.48	3.64	1.68	1.56	1.85	6.5
Location 2	2.71	3.4	1.38	1.86	1.92	5

historically documented changes that occurred in both the region and Yosemite Valley, in particular (e.g., 1849 California gold rush, Mariposa Battalion, Euro-American occupation in Yosemite Valley).

The almost complete cessation of fire after 1890 coincides with the Yosemite Act of 1890 in which Congress set aside areas surrounding Yosemite Valley and created Yosemite National Park when guardianship of these lands surrounding Yosemite Valley fell to the U.S. Cavalry.

By comparing historic lightning ignitions to dendrochronological fire history and archaeological data, I was able to determine that dendrochronological fire history in Yosemite Valley varies dramatically from historical observed lightning-caused fires. In the absence of a dramatic change in climate and lightning-ignition patterns that have not been documented, it can be stated that the modern lightning-ignited fires cannot account for the fire return intervals observed in the prehistoric and early historic dendrochronological fire record. With the archaeological record showing a spatial and temporal overlap of Native American occupation during this period, the fire regime within Yosemite Valley was the product of the human occupants of the valley.

Comparison to Areas Where Lightning Is Not Restricted

To what extent Native Americans influenced prehistoric fire regimes is often debated; but do we know what a Native American fire regime looks like? Would we know it when we see it? Is there a litmus test that geographers, ecologists, and anthropologists can use to determine the amount of human influence? This study showed two variations that could be used as keys for determining if a prehistoric fire regime was created by Native Americans: 1) the dendrochronological fire his-

tory varied greatly from the historically observed lightning-ignition pattern, and 2) changes in fire regimes corresponded with some of the known archaeological chronologies.

I sought to determine if those two findings could be used as a litmus test by testing them against previously published fire histories. Dendrochronological fire histories do not regularly compare fire regimes to historically observed lightning-ignition patterns or local archaeological chronologies and the common data provided in the literature do not allow for that level of analysis without large amounts of additional data.

The data commonly provided in fire-history studies are a composite MFI and seasonality. So how do MFI and seasonality of Yosemite Valley's anthropogenic fire regime compare to other areas where lightning ignitions probably are not as restricted? A literature review of fire histories in mixed conifer on the western slope of the Sierra Nevada found seven studies to compare against the Yosemite Valley data. Comparisons of composite MFI show that Yosemite Valley's anthropogenic fire signature does not differ strikingly from that in other locations (Table 6). The MFI across the western Sierra Nevada is 1.7–5.7 (range: 1–28 y). Yosemite Valley with 1.92 MFI (range: 1–11 y) is within this range of variability. When compared to the two nearest sites, South fork of Tuolumne River with a composite MFI 1.5 and the South fork of Merced River with a composite MFI 2.13 (A.E. Scholl, Pennsylvania State University, personal communication), Yosemite Valley's anthropogenic fire signature continues to be within the range of variability.

Comparisons of seasonality also reveal that the anthropogenic fire signature of Yosemite Valley is not outside the historic range of variability (Table 7). The different distribution of dormant to latewood scars ob-

Table 4. Two-tailed *t*-test comparison of fire intervals by time period, Yosemite Valley, California.

Time period	<i>P</i>
Prehistoric (AD 1520–1800) vs. Protohistoric (AD 1800–1847)	0.0000*
Protohistoric (AD 1800–1847) vs. Historic 1 (AD 1848–1863)	0.1929
Historic 1 (AD 1848–1863) vs. Historic 2 (AD 1864–1890)	0.9610
Historic 2 (AD 1864–1890) vs. Historic 3 (AD 1891–1944)	0.0001*

* Significant at $P \leq 0.05$.

Table 5. Statistical comparison of village type by time period, Yosemite Valley, California. Two-tailed *t*-tests for MFI difference were not significant ($P \leq 0.05$).

Time period	<i>P</i>	Sap-pah'sam-mah vs. Kis'-se	
		Years with synchrony of $\geq 10\%$ samples scarred	
AD 1520–2004	0.8146		
Late Prehistoric 3 (AD 1700–1800)	0.8576	1775, 1783, 1800	
Protohistoric (AD 1800–1847)	0.3171	1841	
Historic 1 (AD 1848–1863)	0.9102		
Historic 2 (AD 1864–1890)	0.8402	1864	

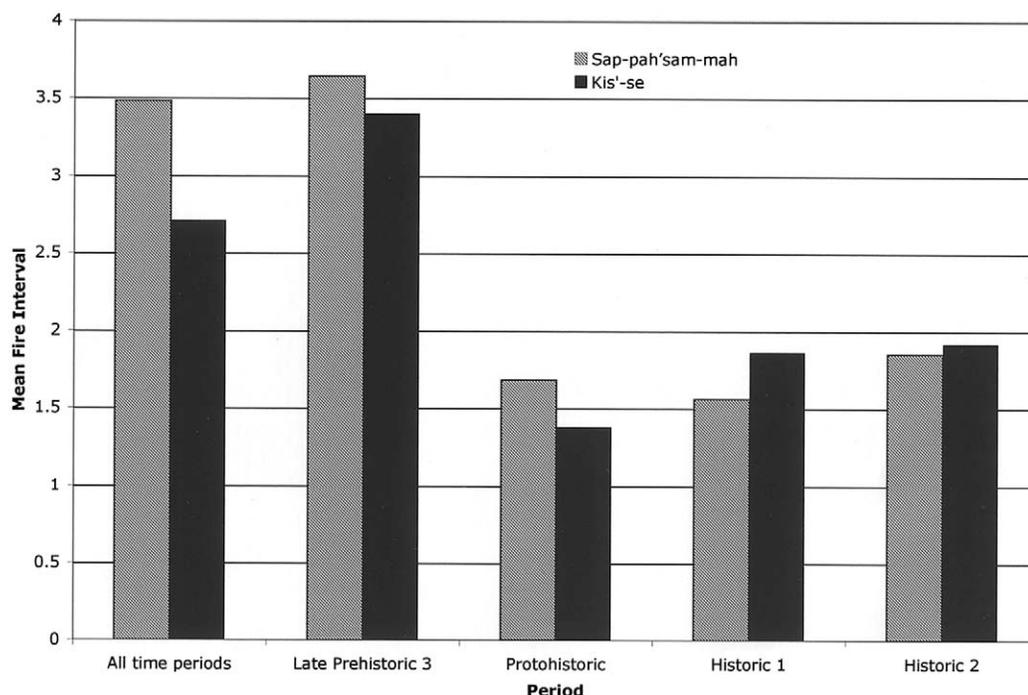


Fig. 5. Mean fire return interval (y) by period for the villages of Sap-pah'sam-mah and Kis'-se, Yosemite Valley, Yosemite National Park, California.

served in Yosemite Valley and other studies is consistent with latitudinal gradients of climate.

Comparison of the anthropogenic fire signature in Yosemite Valley to other studies within the Sierra Nevada shows that Native American fire regimes are not necessarily outside the range of variability of other fire-history studies within the Sierra Nevada. The similarity may be due to a high degree of Native American influence on fire regimes outside of Yosemite Valley, a dependence of fire spread on fuel and climatic events regardless of ignition source, or Native Americans mimicking lightning-fire patterns, or it may be related to data sources and sampling methods. The reasons for the similarity in fire regimes cannot be determined without further research.

Barrett et al. (2005:32) stated “lightning fires . . .

were well capable of maintaining most fire regimes in the West.” In the case of Yosemite Valley, this is not true: lightning fires could not maintain the prehistoric and historic fire regimes seen in the dendrochronological fire history. Whether this can be said for the rest of the West remains to be tested using site-specific human-occupation data. Most fire-history studies are not designed to determine source of ignition or differentiate between potential amount of lightning and anthropogenic ignitions. Currently, we do not even know what a definitive anthropogenic signature looks like and we do not know if or how much it varies over space, time, or how much it is based on human intent and needs.

We know that Native Americans lived in discrete locations and various times. The similarity of Yosemite

Table 6. Composite mean fire interval (MFI) for locations in Yosemite Valley, California.

Location	Study area (ha)	No. of trees sampled	Composite MFI for study area (y)	Composite MFI range (y)	Source
Blodgett	5	46	4.7	4–28	Stephens and Collins 2004
Pilot Creek	15	15	5.7	3–18	Stephens and Collins 2004
South fork of Tuolumne	2,100	209	1.5	1–16	A.E. Scholl, Pennsylvania State University, personal communication
Yosemite Valley	30	57	1.92	1–11	Gassaway 2005
South fork of Merced	1,625	69	2.13	1–28	A.E. Scholl, Pennsylvania State University, personal communication
Southern-central Sierra Nevada, Yosemite National Park	20–50		2–3	1–25	Swetnam et al. 1998, Stephens and Collins 2004
Sequoia Kings Canyon National Park, Redwood Creek	1,030	37	2.1	1.73–2.35	Kilgore and Taylor 1979
Bearskin Creek	770	183	1.7	1.55–1.86	Kilgore and Taylor 1979
Mountain Home Demonstration State Forest	20–50		3–5	1–12	Swetnam et al. 1998, Stephens and Collins 2004

Table 7. Fire seasonality, western slope of Sierra Nevada, California.

Location	Season ^a (%)					Source
	D	L	EEW	MEW	LEW	
Blodgett	21	79	NR	NR	NR	Stephens and Collins 2004
Pilot Creek	21	79	NR	NR	NR	Stephens and Collins 2004
South fork of Tuolumne	51	24	NR	NR	NR	A.E. Scholl, Pennsylvania State University, personal communication
Yosemite Valley	48	31.5	7.3	5.2	7.3	Gassaway 2005
South fork of Merced	38	32	NR	NR	NR	A.E. Scholl, Pennsylvania State University, personal communication
Southern-central Sierra Nevada, Yosemite National Park	23	54	1	4	18	Swetnam et al. 1998, Stephens and Collins 2004
Mountain Home Demonstration State Forest	20	61	0	1	16	Swetnam et al. 1998, Stephens and Collins 2004

^a Abbreviations: D, dormant; EEW, early earlywood; L, latewood; LEW, late earlywood; MEW, middle earlywood; NR, not reported.

Valley's anthropogenic fire signature to other fire histories may indicate that mean fire return intervals and seasonality cannot be used alone to determine the amount of lightning versus anthropogenic intervention within a fire regime. In order to determine the amount of influence Native Americans and lightning had on fire regimes, studies have to either explicitly or quantitatively determine ignition sources and use site-specific ethnographic and archaeological data.

Without additional studies that take into account the site-specific ethnographic and archaeological data, our understanding of anthropogenic fire use will be fraught with subjective opinion and bias. We need to test our assumptions.

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THE HISTORIC FIRE REGIME ON THE EDGE OF THE PRAIRIE: A CASE STUDY FROM THE CROSS TIMBERS OF OKLAHOMA

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ABSTRACT

Changes in the historical disturbance regime because of Anglo-American settlement and forced removal of Native Americans have altered vegetation composition and structure of forests, particularly in mesic ecosystems. However, xeric forests have gone largely unstudied, even though they may contain some of the largest tracts of remnant old-growth forests in eastern North America. Using dendrochronological techniques, we reconstructed fire, drought, and vegetation dynamics of an old-growth forest in the Cross Timbers region of Oklahoma. We tested predictions that fires would 1) be influenced by topography, 2) have decreased in frequency since Anglo-American settlement, 3) increase in frequency during drought events, and 4) be followed by pulses of hardwood recruitment. Support for our first prediction included higher fire frequency on the most southwest-facing aspect. We found no evidence to support prediction 2; fires became more frequent following Native American and Anglo-American movement into the area. Our third and fourth predictions were both supported; large-scale fire events occurred following periods of prolonged drought and 57–100% of oak (*Quercus*) recruitment followed a fire event. In contrast, eastern redcedar (*Juniperus virginiana*), an invasive native species, recruited during fire-free periods. Our results indicate that the sequence of drought followed by fire is an important process in long-term maintenance of forest conditions in this oak ecosystem. We attribute the increase in fire frequency after settlement to increasing local anthropogenic ignitions. The local effect of settlement on the fire regime (i.e., increased fire frequency) seems to contrast with the regional effects of settlement (i.e., decreased fire frequency).

keywords: Anglo-American settlement, Cross Timbers, drought, fire regime, oak recruitment, Oklahoma, stand dynamics.

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INTRODUCTION

Fire has influenced vegetation patterns of forests in eastern North America for millennia (Clark et al. 1996, Delcourt and Delcourt 1998, Bonnicksen 2000). Removal of Native Americans and the onset of Anglo-American settlement have led to changes in the fire regime in many eastern North American forests (Cutter and Guyette 1994, McClenahan and Houston 1998, Ruffner and Abrams 2002). The early days of settlement often brought a temporary increase in fire fre-

quency due to population pressures related to land use (Ruffner and Abrams 2002). However, by the early 1900s, fire frequency was largely reduced from pre-settlement levels, contributing to a decreased oak (*Quercus*) component, particularly on mesic sites where competition from fire-sensitive species is intense (Abrams et al. 1995, Delcourt and Delcourt 1998).

In xeric oak woodlands and savannahs, reductions in fire frequency have led to an increase of invasive plant species, changes in stand structure, and decrease in spatial heterogeneity across the landscape (Jenkins and Rebertus 1994, Bóo et al. 1997, Heikens 1999, Fuhlendorf and Engle 2004). However, few studies have linked the changes in fire frequency to human population changes and to subsequent effects on vegetation response in these ecosystems. Studies that have

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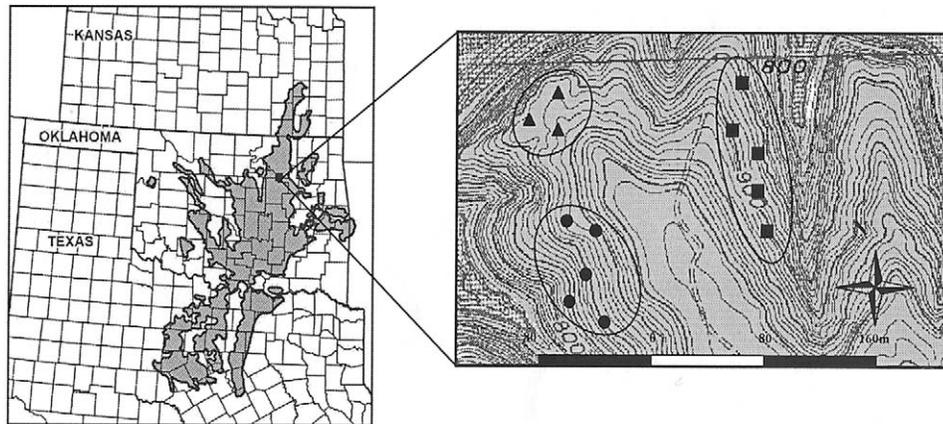


Fig. 1. The Cross Timbers region (adapted from Küchler 1964) with approximate location of Keystone Ancient Forest Preserve (inset) and location of plots in each stand (● = Southwestern slope stand; ■ = Northeastern slope stand; ▲ = Bench stand). Copyright 2005 by Natural Areas Association. From *Natural Areas Journal*, by S.L. Clark, S.W. Hallgren, D.W. Stahle, and T. Lynch, "Characteristics of the Keystone Ancient Forest Preserve, an old-growth forest in the Cross Timbers of Oklahoma." Reprinted by permission of Natural Areas Association.

reconstructed fire history in xeric oak ecosystems have generally been confined to the Ozarks (Guyette and Cutter 1991, Cutter and Guyette 1994, Guyette et al. 2002). These studies have found that fire frequency was generally reduced following Anglo-American settlement and was highest during occupation by the original Native American habitants.

The Cross Timbers region of Oklahoma has gone largely unstudied, but is an important region from a biological and historical perspective (Stahle and Hehr 1984, Therrell and Stahle 1998, Hoagland et al. 1999). The region was occupied by Osage Indians from the 1700s until the 1830s, when bands of Cherokee, Creek, Seminole, Choctaw, and Chickasaw tribes were forced to leave their homes to the southeast to occupy the new Indian Territory in present-day Oklahoma (Wyckoff 1984). Although many explorations took place in the 1800s (Foreman 1947, Albert and Wyckoff 1984), the Cross Timbers was largely unoccupied by Anglo-Americans until 1889 when settlement became legal (Wyckoff 1984).

In this study, we examine how topography, changes in human demographics, and climate influenced the fire regime of an old-growth oak forest in the Cross Timbers region of Oklahoma. Our specific objectives were to 1) reconstruct the fire history of this site through dendrochronological dating of fire scars, 2) examine the effects of topography and climate on fire occurrence and spread, and 3) determine the relationship between tree recruitment patterns and fire events. We had four research hypotheses: 1) fire frequency would be influenced by topography, 2) fire frequency has decreased since Anglo-American settlement, 3) fire events were more likely to occur during periods of prolonged drought, and 4) fire promoted oak recruitment.

STUDY AREA

The study area was part of The Nature Conservancy's Keystone Ancient Forest Preserve (KAFP).

The KAFP lies in the northern Cross Timbers region (Küchler 1964), approximately 32 km west of Tulsa, Oklahoma, in southern Osage County at the confluence of the Cimarron and Arkansas rivers (Figure 1). The region is characterized as a mosaic of xeric oak woodlands, savannas, and prairie openings scattered throughout approximately 4.8 million ha, with nearly half of its area in Oklahoma (Küchler 1964, Hoagland et al. 1999). Average annual precipitation is 101 cm, and average winter and summer temperatures are 3° and 26°C, respectively (Oklahoma Climatological Survey 2002). The KAFP has been characterized as old growth because of the presence of post oak (*Quercus stellata*) and eastern redcedar (*Juniperus virginiana*) trees greater than 300 and 500 y old, respectively, and the absence of anthropogenic disturbances (Therrell and Stahle 1998). Elevation ranges from 251 to 304 m; topography is moderately to steeply sloping with underlying bedrock of shale and sandstone. Soils are a Niotaze-Darnell complex, with a sandy loam surface layer (Bourlier et al. 1979).

Previous research indicated that topography was the primary factor in identifying distinct forest vegetation communities (Roe 1998). Thus, we delineated three stands based on aspect and elevation. The 11-ha Southwestern slope stand was located on the steep upper slopes (slope = 17%) with a southwestern aspect. The 14-ha Northeastern slope stand was located on steep lower slopes (slope = 20%) with a northeastern aspect and had the lowest elevation. The 5-ha Bench stand was located on relatively level terrain (slope = 6%) just north of the main ridgetop.

METHODS

Field Sampling

We randomly placed three to five 55 × 55-m plots within each stand (Figure 1), and we randomly sampled 98 saplings (>1 m height, <10 cm diameter at breast height) from five species for age structure anal-

ysis. We obtained cores or cross sections within 10 cm of the root collar. The number of saplings selected within a species was consistent with species' relative densities (Clark et al. 2005). By sampling smaller trees at the base, we could examine recruitment response to recent fire events at a near-annual resolution (Villalba and Veblen 1997); all cores and cross sections either hit pith or were estimated to be within 1 or 2 y of pith.

Fire scars from dead tree cross sections were used to reconstruct the fire history of the KAFP. We obtained 11, 8, and 8 cross sections from dead tree logs or snags within the Southwestern slope, Northeastern slope, and Bench stands, respectively. We also obtained 2 cross sections along the main ridgeline, just outside of the designated stands' boundaries. For downed logs, cross sections were taken as close to the log's base as possible (<30 cm from base) to improve chances of detecting injury from fire (Guyette and Cutter 1991, Smith and Sutherland 1999). We similarly obtained wedges from the base of standing snags. To provide a fire history of most recent fire events, we documented fire scars on sapling cross sections. We analyzed 13, 3, and 2 saplings that contained fire scars from the Southwestern slope, Northeastern slope, and Bench stands, respectively. Including dead trees and saplings, we collected 46 samples for the fire scar analysis from post oak ($n = 23$), blackjack oak (*Q. marilandica*; $n = 10$), black hickory (*Carya texana*; $n = 9$), and Shumard's oak (*Q. shumardii*; $n = 4$).

Dendrochronological Techniques

Sapling and dead tree samples were dried and sanded with progressively finer grades of sandpaper (100–400 grit) and cross-dated using a regional post oak chronology available in the International Tree-Ring Data Bank (ITRDB) (NOAA 2004). Cross-dating provides confidence that samples were dated to the exact calendar year (Douglass 1941, Stokes and Smiley 1996). Cross-section and sapling tree-ring widths were measured to the nearest 0.001 mm with a Velmex stage micrometer. Portions of cores containing rot were not measured. Many cross sections exhibited eccentric ring growth and were measured along the radius that was intermediate in ring width. We were neither able to cross-date nor to measure eastern redcedar samples because of occurrence of missing rings and false rings, and ages of these trees should be considered ring counts and not exact ages (Schweingruber 1988).

We identified fire scars on the dead tree samples and dated them to an exact calendar year. Fire scars are defined as wounds resulting from cambial death due to excessive heating or scorching (Smith and Sutherland 1999). All fire scars were associated with abnormal vessel formation and production of callus tissue, i.e., undifferentiated cells that contain little lignin (Smith and Sutherland 1999), which resulted in wide growth rings in the year of fire and sometimes several years following the fire. Fire scars from two known and documented fire events at the KAFP in 1994 and 1996 were also used as reference for identifying fire scars in previous years. Fire scars sometimes produced

woundwood ribs at the margins of the fire scar (Smith and Sutherland 1999), particularly if the fire injured a large portion of the tree's circumference. However, fires sometimes caused relatively localized injury and did not cause formation of woundwood ribs. Fire scars generally occurred during the dormant season that spans two calendar years (i.e., after the end of the previous year's growing season). The fire scars were assigned the year in which the tree first showed a wounding response to the fire (i.e., the year containing the earlywood immediately following the fire scar).

Fire Scar Analysis

Fire scar analysis was conducted using FHX2 software (Grissino-Mayer 2001). Years were assigned as recorder or null years, and only recorder years are included in the analysis. Recorder years include the time period from the first year the tree recorded a fire to the year of the tree's outer ring. Once a tree is scarred, it becomes more susceptible to injury by subsequent fires and years between two fire events are, therefore, considered recorder years. Years from the inner ring to the first fire event were considered null years because proof of fire events were not being recorded, even though the tree may have been experiencing fire during this time. Null years included prolonged periods (>100 y) between two fire events; if the tree completely healed the wound from the previous fire event, proof of fire history information does not exist and was therefore not included in the sample (Grissino-Mayer 2001). Null years also included rings that were unreadable due to rot or because fire burned off portions of the wood.

We performed fire scar analyses for each stand and across stands; separate analyses were also conducted for all fire events, and for moderate-scale fires, defined as fires that scarred $\geq 25\%$ of sample trees. We calculated the fire return interval for each stand and across stands, defined as the number of years between two consecutive fire events and is a measure of fire frequency. We determined the fire return interval data were not normally distributed according to a Kolmogorov–Smirnov test for goodness of fit ($P < 0.05$). We therefore used the median fire return interval as an indicator of fire frequency, which is a better descriptor of the central tendency of a data set than the mean when the data are not normally distributed.

We calculated a fire index to determine how fire effects have changed through time (Guyette and Cutter 1991). The index compensates for the effects of sampling size bias; generally, sample size will increase with more recent fire history, and number of fire scars detected will increase with increasing sample size. We calculated the index by dividing the number of fire scars in each 10-y time interval by the number of trees in the record for the same interval.

We performed temporal change analysis to determine if fire frequency has changed because of the movement of displaced Native Americans into the region and the onset of legal Anglo-American settlement. A *t*-test was used to determine differences in the

mean fire return interval and the percentage of trees scarred between a control period and two test periods. The control period was from 1772 to 1830 and represents time from the beginning of the tree-ring chronology to the beginning of Native American movement into the area from their native homelands to the east. The first test period was from 1830 to 1889 and represents the time period of occupation by displaced Native Americans until legal Anglo-American settlement. The second test period was from 1889 to 2002 and represents the time period from legal Anglo-American settlement to the end of the tree-ring chronology. The data were first transformed to the standard normal distribution to apply the *t*-test. A probability value of 0.05 was chosen to indicate significance.

Analysis of Climatic Effects

Annual instrumental Palmer drought severity indices (PDSIs) from 1900 to 2002 and annual reconstructed PDSIs for 1770 to 1899 were obtained by averaging data from grid points 178, 178, 193, and 194 available from the NOAA Paleoclimatology Program, the National Geophysical Data Center, and the National Climatic Data Center (<http://www.ncdc.noaa.gov/paleo/pdsi.html>; Cook et al. 2004). Monthly PDSI values from 1895 to 2002 were obtained using data from the National Climatic Data Center for Oklahoma, Division 5 (NOAA 2004). The PDSI ranges from -4 (extreme drought) to $+4$ (extreme wet) and represents several environmental variables that affect tree growth, including drought duration, soil water-holding capacity, temperature, and precipitation (Palmer 1965). We adjusted the annual reconstructed values by multiplying by the ratio of the two variances:

$$rPDSI \times \left(\frac{\sigma_{iPDSI}}{\sigma_{rPDSI}} \right) = adjPDSI,$$

where *rPDSI* = the annual reconstructed value, *iPDSI* = annual instrumental value, and *adjPDSI* = the annual adjusted reconstructed value.

We conducted a superposed epoch analysis using Program EVENT (Holmes and Swetnam 1994) to determine if drought occurred prior to and during a dated fire event. The EVENT program computes average values for an independent variable during a window of time surrounding an event year (i.e., in this study, an event year was a year of a fire) and compares average values of the independent variable to randomly generated predicted values. The predicted values represent 1,000 random simulations generated for each event. A departure value is then calculated for each year in the event window that represents the actual value minus the predicted value. A statistically significant departure value is detected when the actual value is greater than or less than the 95% confidence limits (based on a normal distribution) of the predicted value.

We conducted the analysis for two types of fire events: a moderate-scale fire, defined as fires that scarred $\geq 25\%$ of sample trees; and a large-scale fire, defined as a fire that scarred at least 50% of sample trees or a fire that burned in all three stands. The latter

Table 1. Median fire return interval (FRI) with coefficient of variation (CV) within and across stands for all fire events and for moderate-scale fires at the Keystone Ancient Forest Preserve (KAFP), Oklahoma.

Variable	Stand			
	Southwestern slope	Northeastern slope	Bench	KAFP
All fires				
FRI	2.5	6.0	3.5	2.0
CV	0.9	1.1	1.7	0.9
Minimum interval	1.0	1.0	1.0	1.0
Maximum interval	18.0	45.0	62.0	15.0
Moderate-scale fires				
FRI	4.0	6.0	4.5	7.0
CV	0.9	1.1	1.4	1.2
Minimum interval	1.0	1.0	1.0	1.0
Maximum interval	20.0	45.0	62.0	49.0

type of fire event would represent only the most widespread fires at the study site. We conducted the analysis for annual PDSI values from 1770 to 2002 using the reconstructed and instrumental values as described above. We also conducted the analysis for each season within a year by averaging monthly PDSI values from the current year as follows: January–March for winter, April–June for spring, July–September for summer, and October–December for fall. The analysis for seasonal PDSI values could only be conducted for years after 1895 when instrumental data were available.

RESULTS

Topoedaphic and Temporal Patterns of Fire Frequency

The median fire return interval from 1772 to 2002 for all fire events was 2.0 and was 7.0 for moderate-scale fires (Table 1). Approximately 40% of fires scarred only one tree. Fires occurred most frequently in the Southwestern slope stand and least frequently in the Northeastern slope stand. The Bench stand had the longest single fire-free interval (62 y). A total of 77 fires were recorded, with total number of fires and sample size increasing over the length of the chronology (Figure 2). We documented 24 moderate-scale fire events and 8 large-scale fire events (Table 2).

The fire index value increased slightly over the last 150 y and was lowest during the 1830s, with no fires reported during that decade (Figure 3). The fire return interval for all fire events did not change significantly with the onset of Native American displacement into the area according to the temporal change analysis (Table 3). However, the fire return interval was significantly shorter following Anglo-American settlement compared with the control period. Conversely, moderate-scale fire return interval increased from 4.9 during the control period to 15.0 following Anglo-American settlement, but this difference was not significant. The temporal change analysis performed on percentage of trees scarred differed from fire return interval analysis. The percentage of trees scarred for all fire events was significantly lower during the two test pe-

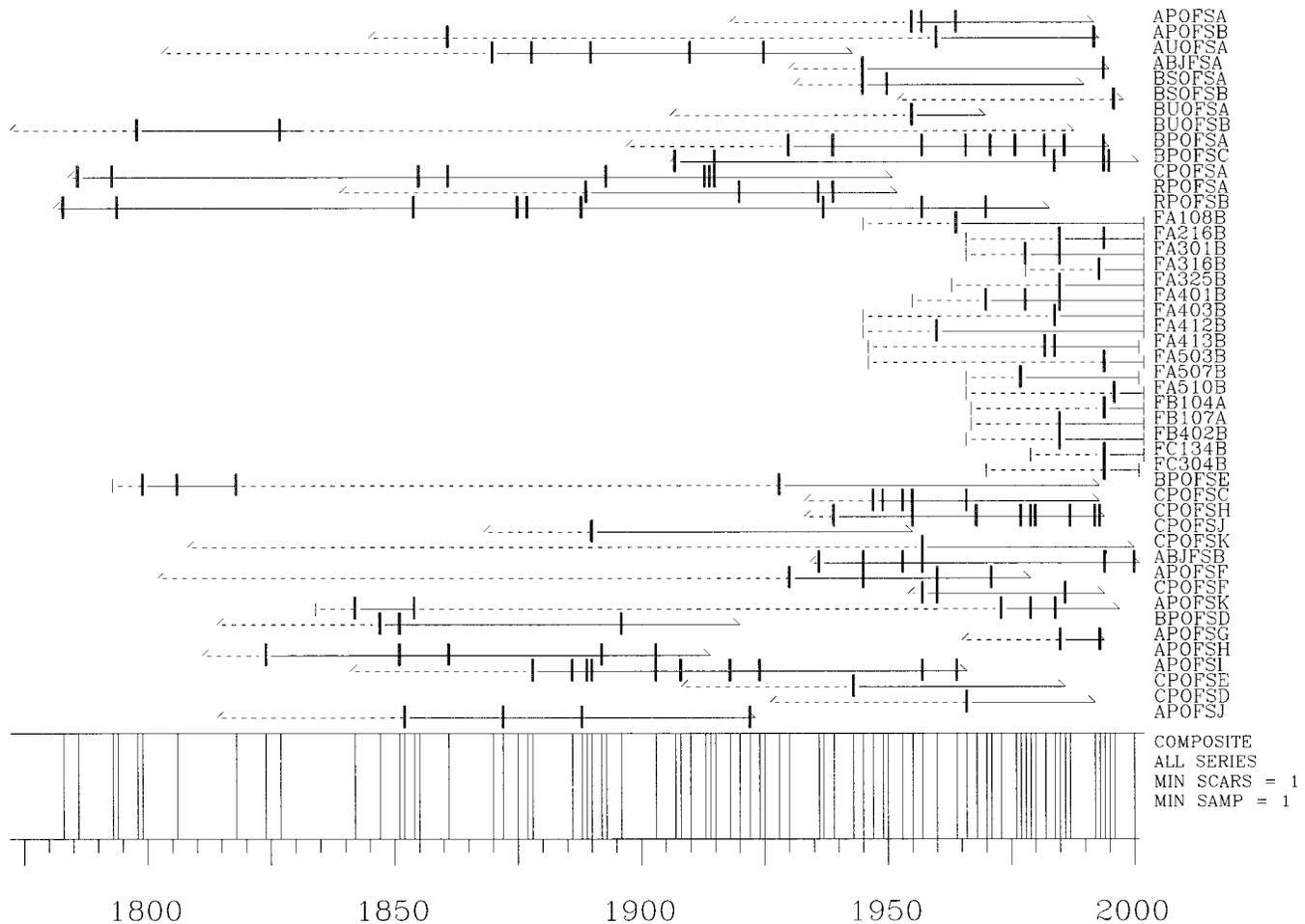


Fig. 2. Fire chronology from 1772 to 2002. Horizontal lines represent individual sample trees (author's nomenclature) and vertical dashes represent fire events; dashed horizontal line represents period not included in analysis and solid horizontal line represents period included in fire analysis. Bottom bar is master fire chronology for all samples in the Keystone Ancient Forest Preserve, Oklahoma.

riods of Native American displacement and Anglo-American settlement compared with the control period. However, no differences were detected for moderate-scale fire events.

Effects of Climate

Neither moderate-scale nor large-scale fire events were related to annual PSDI values, but fire did appear to be dependent on drought during a particular season. The superposed epoch analysis indicated that summer and fall PDSI values preceding a dated fire event were

Table 2. Moderate-scale and large-scale fire events at the Keystone Ancient Forest Preserve, Oklahoma.

Moderate-scale fire			Large-scale fire
1783	1824	1889	1783
1786	1827	1890	1786
1793	1842	1939	1793
1794	1851	1945	1794
1798	1854	1955	1861
1799	1861	1957	1955
1806	1878	1985	1957
1818	1888	1994	1994

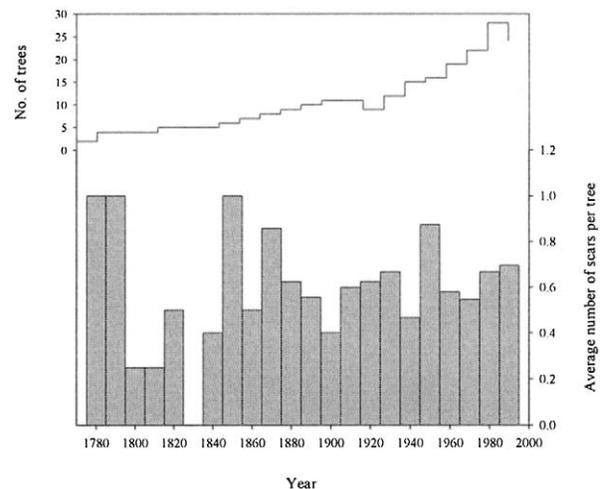


Fig. 3. Fire index values (number of scars per tree) per 10-y interval, Keystone Ancient Forest Preserve, Oklahoma. The top graph represents sample size used in our analysis.

Table 3. Comparison in differences in fire return interval and percent trees scarred (standard error in parentheses) between control time period (1772–1830), and displaced Native American period (1830–1889) and Anglo-American settlement test period (1889–2002).

Variable	Control period (1772–1830)	Test period (1830–1889)	Test period (1889–2002)
Fire return interval			
All fires	4.9 (1.2)	3.4 (0.7)	2.1 ^a (0.2)
Moderate-scale fires	4.9 (1.2)	7.8 (2.3)	15.0 (6.6)
Percent trees scarred			
All fires	40.8 (7.5)	24.5 ^a (2.6)	14.2 ^a (1.2)
Moderate-scale fires	40.8 (7.5)	32.9 (3.5)	31.4 (2.6)

^a t-test with control period was significant ($P < 0.05$).

contributing factors in the occurrence of large-scale fires (Figure 4). Moderate-scale fire events showed the same trend as large-scale fires, but departure from normal PDSI values was not significant.

Recruitment Dynamics

Recruitment pulse of hardwood species predominately occurred following fire events in each stand (Figure 5). Sapling recruitment for the four hardwood species occurred within 2 y following a fire 61–90% of the time, depending on stand. This pattern was greater than expected by chance alone, given the length of the sapling age structure. Fire appeared to have the most effect on recruitment in the Northeastern slope stand and the least effect in the Bench stand. A large recruitment pulse followed the 1945 and 1964 fires in the Southwestern slope stand and followed 1966 in the Northeastern slope and Bench stands. Recruitment response to fire was similar among species, except for eastern redcedar. While some saplings were able to withstand fire, the majority of eastern redcedar recruited during prolonged fire-free periods (>15 y), particularly in the Bench stand, where it had the highest abundance and was increasing in density.

DISCUSSION

Differences across the Topoedaphic Gradient

We found support for our first hypothesis that fire frequency was influenced by topography. However, inferences from our results are speculative due to the lack of replication of different stands. Landscape has been shown to have a strong local effect on a fire regime, due to effects on insolation and fuel moisture (Bergeron 1991). The Southwestern slope and Bench stands would have the highest levels of insolation, due to reduced canopy cover and southerly exposure (Clark et al. 2005) and would therefore have the highest probability of fire ignition and spread (Pyne et al. 1996), as our results support. In contrast, the Northeastern slope stand would have the highest levels of fuel moisture and less probability of fire. A drainage to the east (Figure 1) may also serve as a natural fire-break.

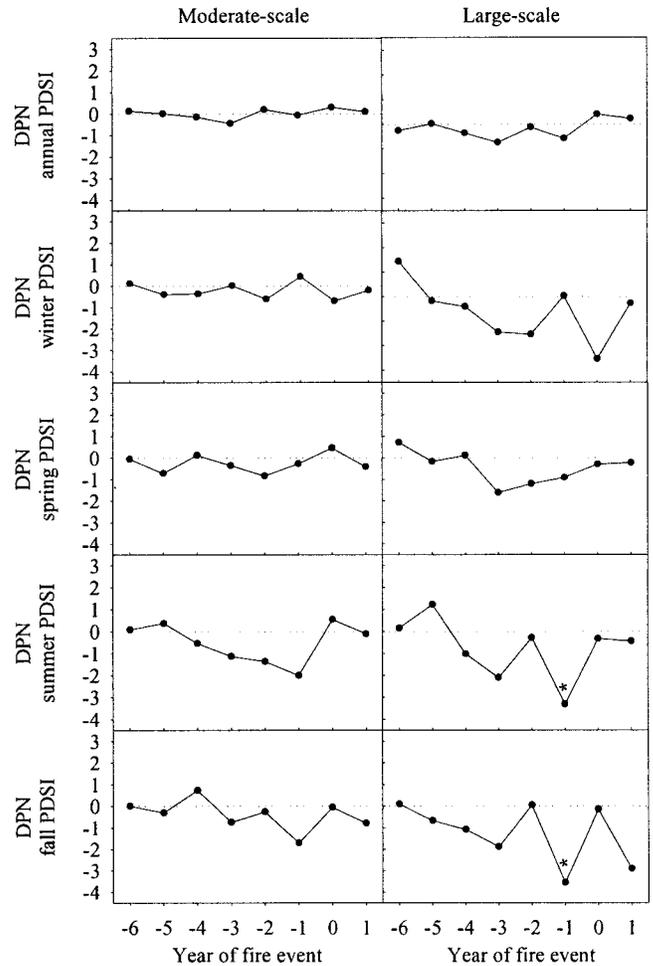


Fig. 4. Departure from long-term normal (DPN) Palmer drought severity index (PDSI) for fire event (year 0) and years surrounding fire event for annual and seasonal PDSI values and for moderate- and large-scale fire events, Keystone Ancient Forest Preserve, Oklahoma. Asterisks indicate years in which departure from normal PSDI was significant ($P < 0.05$).

Temporal Changes in the Fire Regime

We did not find support for our second hypothesis that fire frequency has decreased since Anglo-American settlement. Fire frequency increased after Anglo-American settlement, differing from numerous studies conducted in oak forests throughout the midwestern and eastern United States (Guyette and Cutter 1991, Cutter and Guyette 1994, Abrams et al. 1997, Orwig et al. 2001, Shumway et al. 2001). The fire return interval at the KAFP was shorter than those reported in Missouri Ozark forests (Guyette and Cutter 1991, Cutter and Guyette 1994, Batek et al. 1999) and in oak gallery forests of northeastern Kansas (Abrams 1985) of similar or larger size. The lack of significant change in fire frequency with the period of Native American displacement is attributed to the fact that the actual influx of Native Americans during this time at this particular site was relatively low (Wyckoff 1984). Human populations did not increase dramatically until after the Osage Indian lands were allotted after statehood in 1907, coinciding with the first oil explorations in

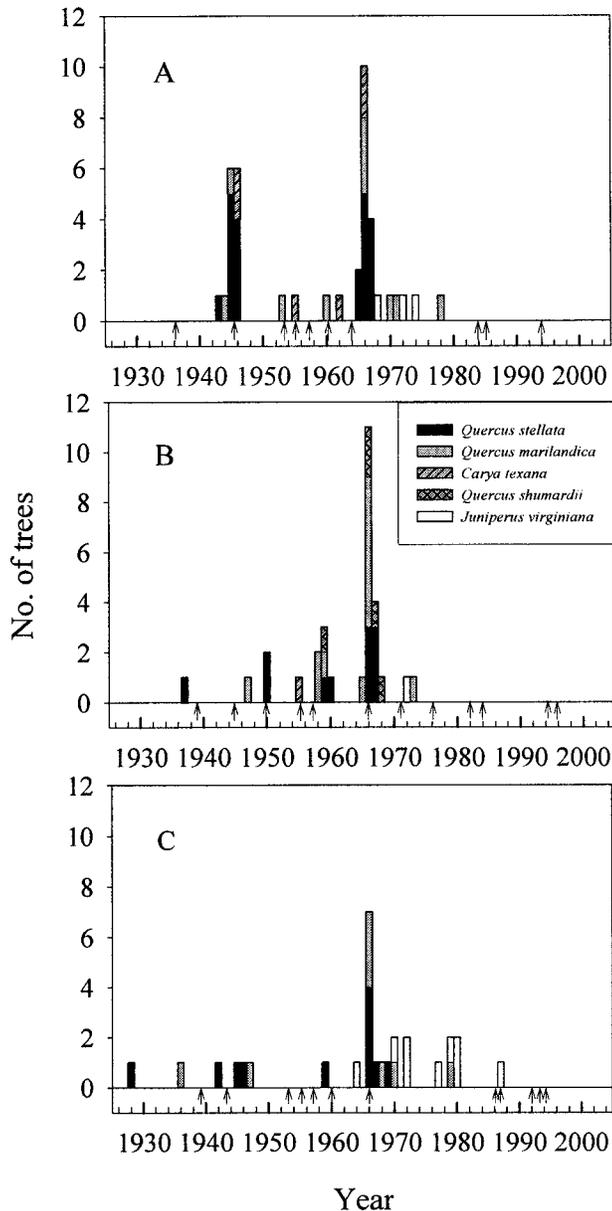


Fig. 5. Age structure of saplings in the (A) Southwestern slope, (B) Northeastern slope, and (C) Bench stands, Keystone Ancient Forest Preserve, Oklahoma. Arrows represent moderate-scale fire events.

the area that continue to the present day. The potential for anthropogenic ignitions increases as human population density increases, as supported by our results (Guyette and Cutter 1991, Guyette et al. 2002). Additionally, construction of a major highway in the mid-1900s and the frequent use of fire by local farmers to clear fields may have contributed to the increase of fire frequency during the past 100 y.

Sample size bias may be artificially increasing fire frequency. As sample size increases over time, the possibility of finding fire scars also increases, thereby biasing our results. This potential bias is also supported by the fact that the percentage of trees scarred decreased over time. However, other studies also had similar potential for bias due to sample size (Guyette

and Cutter 1991) and did not show an increase in fire frequency, giving some support to an actual increase in ignition because of increased human population pressure.

The fire return interval for moderate-scale events would have increased if we had included null years prior to the first fire event and periods between two prolonged fire events in the analysis. Although some suggest these periods represent real fire-free intervals (Baker and Ehle 2001), years in which fire history information is not known should not be included in the analysis because no scientific information is available for that time period.

Drought Influences on Fire

We found support for our third hypothesis that climatic conditions affect fire frequency, but only for large-scale fire events. We speculate that while ignition of fire is likely independent of climate, the ability of fire to spread across the landscape may be partially dependent on climatic conditions, particular during the previous summer and fall. This can best be seen with the droughts of the late 1850s and mid-1950s, when PDSI levels remained below normal for several consecutive years (data not shown). Consequently, large-scale fires occurred (Table 2) and fire index values were high (Figure 3) during both of these periods.

The analysis of seasonal effects on fire in the Cross Timbers is lacking in many studies and may help explain the conflicting results among past studies, even within the same climatic region. Fire was not related to drought in the Missouri Ozarks (Cutter and Guyette 1994), but a relationship was found in a nearby post oak savanna (Guyette and Cutter 1991). These previous studies did not examine seasonal effects of climate on fire, which was shown to be important in this study.

Oak species are particularly sensitive to climate, producing narrow rings (Stahle and Hehr 1984) and exhibiting high mortality during severe droughts (Rice and Penfound 1959, Jenkins and Pallardy 1995, Olano and Palmer 2003). Changes in fuel moisture and availability related to drought could be affecting fire spread. Drought increases fine fuel loads due to death of live plant material and reduces fuel moisture in large fuels (Pyne et al. 1996). Thus, fuel unavailable to burn during normal climate conditions becomes available during dry periods, facilitating fire spread and intensity (Engle et al. 1989, Bidwell and Engle 1991, Pyne et al. 1996).

Sapling Recruitment Response to Fire

We found evidence to support our fourth prediction that fire promoted oak recruitment into the sapling layer. Our results support theories that oaks exhibit a "bottleneck" effect, whereby recruitment from seedlings to saplings is encouraged by periodic disturbances (Johnson 1992, 1993; Johnson et al. 2002). Evidence of the bottleneck effect in this forest is also supported by previous studies at the KAFP that showed seedling populations were high (approximately 6,000–13,000 trees ha⁻¹), while sapling populations were

comparatively low (approximately 1,500–2,000 trees ha⁻¹) (Clark et al. 2005). Adaptations such as thick bark and deep rooting provide oaks with resistance to fire (Hengst and Dawson 1994, Abrams 1996), whereas fire promotes prolific sprouting of oaks, particularly in the more xerophytic species (Penfound 1968, Crow et al. 1994, Johnson et al. 2002, Clark and Hallgren 2004). Black hickory was also encouraged by fire in this forest, indicating that this species may have similar adaptations and recruitment dynamics as the oak species.

In contrast to hardwood species, fire restricted the establishment of eastern redcedar, a species that has been invasive in prairie and glade communities throughout the Midwest (Bragg and Hubert 1976, Engle et al. 1996). This species appeared to recruit under fire-free conditions and would likely increase in these stands in the absence of fire (Beilmann and Brenner 1951). Sapling population structure suggests that this species is increasing in the Bench stand, with no recruitment prior to 1964. This finding is further substantiated by the fact that large overstory species were relatively rare in this stand (Roe 1998, Clark et al. 2005).

CONCLUSIONS

On a local scale, Native American displacement and Anglo-American settlement had a positive effect on fire frequency and oak recruitment, in contrast to findings from the majority of dendroecological studies in eastern North America. On a regional scale, Anglo-American settlement has increased eastern redcedar invasion due to the seed rain from surrounding invaded fields and prairies (Engle et al. 1996). Fire appeared to limit the early establishment of eastern redcedar, a species that can negatively affect oak recruitment due to shading and competition for water (Ormsbee et al. 1976; Wittwer 1985; Engle et al. 1987, 1996; Oswald et al. 1996), but once established, this species was able to resist fire.

Despite changes in human demographics, drought is the underlying force of this ecosystem; fire frequency of large-scale fires increased after drought, which helps promote sprouting of hardwood species through top-kill of live saplings. Drought can also directly cause mortality of mature trees (Rice and Penfound 1959, McGee 1984, Olano and Palmer 2003) and thus directly create or maintain canopy gaps suitable for seedling recruitment.

Stand dynamic processes of this forest include interactions among human demographics, climate, and fire. The management of old-growth upland oak forests should include a burning regime that incorporates drought and local human impacts in the decision-making process. A special management consideration may be removal of invasive species, such as eastern redcedar, particularly where invasion is favored by specific site conditions. Although fire-history information and analysis in this study was limited by the number of trees sampled and recording fire history, our results

can provide a building block in understanding the stand dynamics of xeric oak forests in the Cross Timbers. This information will become increasingly important as society simultaneously demands and threatens forest conservation.

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LANDSCAPE CHARACTERISTICS OF SAGEBRUSH-STEPPE/ JUNIPER WOODLAND MOSAICS UNDER VARIOUS MODELED PRESCRIBED FIRE REGIMES

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ABSTRACT

Juniper (*Juniperus* spp.) encroachment into adjacent vegetation has been identified as an ecological and management issue for juniper woodlands. This encroachment has resulted in numerous site- and landscape-level ecosystem changes, such as reduced herbaceous biomass production, altered watershed characteristics, reduced site and landscape diversity, and increased soil erosion. The cause of the encroachment includes direct and indirect fire suppression, livestock grazing, and climatic change. Regardless of cause, the use of prescribed fire has been proposed in many ecosystems as a management technique to address juniper encroachment. The effects of prescribed burning programs on the characteristics of juniper woodland-dominated landscapes and their ability to meet long-term ecological and management objectives have not previously been assessed.

Three 6th-order watersheds were selected on the Owyhee Plateau, Idaho, because of the availability of prescribed fire effects data and other supporting data. The watersheds were classified into potential vegetation types and the area of each was classified into current successional stages utilizing Landsat 7 Enhanced Thematic Mapper (ETM+) and digital elevation model data. Landscape dynamics under four different prescribed fire regimes were modeled for each watershed with the Tool for Exploratory Landscape Scenario Analyses (TELSA). The fire regimes studied were prescribed burning levels affecting 0, 2, 5, and 7% of each watershed's area per decade. Ten simulations of the predicted landscape composition for each watershed under each fire regime were analyzed using FRAGSTATS.

Our results suggest that encroachment of western juniper (*J. occidentalis* ssp. *occidentalis*) is likely to continue into the future if no fire or even limited prescribed burning were implemented. Continued encroachment of western juniper is predicted to result in decreased landscape diversity. While the overall mean patch size within the landscapes decreased, mean patch size of juniper-dominated patches increased. This increase, together with decreases in Simpson's evenness indices, indicate a continued successional trajectory of simplification of watershed composition. Prescribed fire can be utilized to maintain landscape diversity or at least minimize the loss of diversity. Fire occurrence on at least 5% of the watersheds per decade will be required to maintain sagebrush-steppe vegetation at current levels.

keywords: FRAGSTATS, Idaho, *Juniperus occidentalis*, landscape diversity, landscape model, successional model, TELSAs, western juniper.

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INTRODUCTION

Western juniper (*Juniperus occidentalis*) woodlands dominate approximately 3.6 million ha in the northwestern portion of the Great Basin and southern Columbia Basin (Miller et al. 2005). Western juniper has been classified into two subspecies: *J. occidentalis* ssp. *australis*, which occurs in the Sierra Mountains of California; and *J. occidentalis* ssp. *occidentalis*, which occurs in extreme northern California, eastern Oregon, and southwestern Idaho (Vasek 1966, Burkhardt and Tisdale 1969). Western juniper-dominated areas are considered to be the northwestern equivalent of the Intermountain pinyon–juniper zone (West 1999) but do not have an associated codominant pine (*Pinus* spp.). Widespread encroachment of western juniper appears to have accelerated during the last 150 y (Burkhardt and Tisdale 1969, 1976; Miller and Rose 1994; Miller

and Wigand 1994; Miller et al. 2005), which coincides with the period of Euro-American settlement and development of western North America. Research on age-class distributions of western juniper stands indicates that recent expansion began during the period between 1860 and 1880 (Burkhardt and Tisdale 1969, Miller and Rose 1994), slowed during the 1930s and 1940s, and since 1960 has been progressing at an exponential rate (Miller and Rose 1994). The area currently dominated by western juniper in southwestern Idaho has more than doubled since 1960 (Burkhardt and Tisdale 1969).

During the pre–Euro-American settlement period, western juniper is thought to have primarily occurred as dense stands on the more dissected topography or to have occurred as open, savanna-like woodlands on canyon slopes and more regular topography (Burkhardt and Tisdale 1969, 1976; O'Rourke and Ogden 1969; Vasek and Thorne 1977; Miller and Rose 1994; Miller and Wigand 1994). Western juniper has pri-

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marily encroached into many adjacent vegetation types, but the encroachment has been most dramatic in vegetation types with deeper soils (Young and Evans 1981, Eddleman 1987, Miller and Wigand 1994, Miller et al. 1995). These sites include those dominated by mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) steppe, aspen (*Populus tremuloides*) woodlands, and Idaho fescue (*Festuca idahoensis*)/bluebunch wheatgrass (*Pseudoroegneria spicata*) steppe. Encroachment has also occurred into low sagebrush (*Artemisia arbuscula*) steppe; however, the rate has been much lower due to less productive site conditions (Miller et al. 2005).

Causes for encroachment of western juniper are thought to be similar to those of juniper woodlands in general (Bunting 1993, Miller and Wigand 1994). Research has shown that changes in plant competition are probably not a factor in western juniper encroachment because plant composition did not affect rate of establishment (Burkhardt and Tisdale 1976, Eddleman 1987, Miller and Rose 1994). However, heavy utilization of rangelands by livestock could facilitate juniper establishment through secondary effects. The resulting reduction in fine fuel loads due to high forage utilization would decrease fire occurrence (Bunting 1993, Miller et al. 1995) and increase sagebrush seedling establishment (Ellison 1960, Tisdale 1969). Increased sagebrush density provides greater availability of safe sites for juniper seedling establishment (Burkhardt and Tisdale 1976, Eddleman 1987, Miller and Rose 1994, Miller et al. 1995). Miller et al. (2005) found that pre-Euro-American period fire-free intervals (FFI) in western juniper varied depending on local conditions but was usually less than 20 y for woodlands associated with mountain big sagebrush steppe. Young and Evans (1981) estimated, based on the growth rate of young western juniper seedlings, that a fire every 50 y or less would prevent their encroachment in northern California. The encroachment of juniper usually reduces the herbaceous production on the site (Tausch and Tueller 1990) and thereby greatly reduces fire potential (Bunting 1987, Everett 1987). Wildfires may burn only under the most severe weather conditions in dense stands of mature juniper.

Western juniper encroachment into sagebrush steppe or grassland communities decreases herbaceous and shrub biomass production (Vaitkus and Eddleman 1991, Bunting 1993, Miller and Wigand 1994, Miller et al. 1995). A reduction of plant species richness and species diversity has been documented for western juniper and other Great Basin juniper woodlands (Blackburn and Tueller 1970, West et al. 1979, Koniak and Everett 1982, Miller et al. 2000). Bunting et al. (1999) did not find a decrease in species richness. However, species diversity declined due to changes in species equitability of canopy coverage. Sites were increasingly dominated by a single species, western juniper, with associated declines in the canopy coverage of other species.

Studies have shown that fire occurrence can be effective in maintaining early seral vegetation communities associated with juniper woodlands (Miller et

al. 2005). However, little information exists on the proportion of fire on the landscape required to stabilize composition or the effects of fires on landscape characteristics. Our study was designed to address these questions for the Owyhee Plateau in southwestern Idaho. We hypothesized that at some level of fire occurrence within the landscape, the current characteristics (i.e., proportions of cover types and landscape metrics) of the watersheds could be maintained into the future.

STUDY AREA

The Upper Owyhee River basin is located within the southern portion of the Owyhee Plateau in southwestern Idaho (lat 42°30'N, long 116°50'W) and was selected for this study because of the abundance of western juniper-dominated vegetation and availability of auxiliary data. Western juniper forms a noncontinuous woodland zone between 1,450 and 2,100 m in elevation within the Owyhee Plateau. The cool, semi-arid climate is characteristic of the northern Great Basin, with precipitation varying from 25 to 50 cm. Three 6th-order hydrologic code (HUC⁶) watersheds, ranging in area from 6,380 to 6,910 ha, were chosen that had western juniper vegetation on at least 60% of the watersheds' area with a variety of successional stages present. The selected areas include Red Canyon Creek, Smith Creek, and Currant Creek watersheds. Currently, wildfires rarely occur and are suppressed. Prescribed fire has not been commonly applied in these watersheds during the past decade.

METHODS

The potential vegetation type (PVT) classification utilized was developed from a modified classification of the Interior Columbia Basin Ecosystem Management Project (ICBEMP) midscale analysis (Quigley et al. 1996, USDA 1996). PVTs are defined as groupings of habitat types that have similar overstory composition, structure, and environmental requirements, and consequently are broader than habitat types developed for the Columbia Basin. The ICBEMP classification of the watersheds was modified because, for example, many sites that currently supported juniper were not classified as a juniper PVT. We reclassified those areas misclassified to the appropriate PVT taxonomic unit.

Two PVTs of primary interest for this study were defined for the three watersheds: western juniper/mountain big sagebrush and western juniper/low sagebrush woodland. In addition, the following PVTs that covered a smaller portion of the watersheds were defined in order to create a continuous vegetation layer for the watersheds: western juniper-curl-leaf mountain mahogany (*Cercocarpus ledifolius*) woodland, western juniper woodland/rock, aspen woodland, broadleaf riparian woodland, mountain shrubland, low sagebrush steppe, and meadow.

A successional stage classification was developed for each PVT. The classifications were developed by modifying those previously described by Quigley et

Table 1. Successional stage descriptions of the three most prevalent potential vegetation types found within three watersheds in the Owyhee Plateau, Idaho.

Successional stage	Potential vegetation type	
	Western juniper/mountain big sagebrush	Western juniper/low sagebrush
Herbland Shrubland	Shrub cover <5%, herbaceous cover <67% Canopy of low (<50-cm) and/or medium 50- to 200-cm shrubs with a canopy coverage <67%, tree coverage <2%	Shrub cover <5%, herbaceous cover <67% Canopy of low (<50-cm) shrubs with a canopy coverage <67%, tree coverage <2%. Sometimes very open stands of large mature juniper trees are present.
Woodland initiation (Phase 1)	Canopy (<5%) of usually young, sometimes mid-aged junipers present. Juniper is having only minor effects on competition and environment of the site, and the sagebrush community is intact except directly under juniper trees.	Juniper canopy (<3%) of usually young, sometimes mid-aged junipers present. Juniper is having minor effects on competition and environment of the site, and the sagebrush community is intact except directly under juniper trees.
Open young woodland (Phase 2)	Canopy (5–10%) of young and mid-aged junipers present. Juniper beginning to have an effect on the environment of the site. Sagebrush-steppe species declining and sagebrush skeletons often present.	Canopy (3–8%) of young and mid-aged junipers present. Juniper beginning to have an effect on the plant interspace environment of the site. Sagebrush-steppe species declining and sagebrush skeletons present. However, low sagebrush still common in interspaces.
Young multistory woodland (Phase 3)	Canopy (>10%) of young and mid-aged junipers present. Few or no mature junipers present. Sagebrush skeletons often numerous in understory.	Canopy (>8%) of young and mid-aged junipers present. Usually a few mature junipers present. Sagebrush skeletons often present in understory. However, low sagebrush still common in interspaces.
Mature woodland	Overstory canopy >15%, composed of primarily mature individuals (flat-topped trees and wolf lichen [<i>Letharia vulpina</i>] usually present). Few sagebrush remain except in larger openings. Some stands are completely dominated by old mature trees. Other stands may have open canopy of mature trees and a codominant layer of various aged mid-aged junipers.	Overstory canopy >8%, composed of primarily mature individuals (flat-topped trees and wolf lichen usually present). Trees may not necessarily be large sized. Sagebrush usually present in openings except in the denser stands of juniper.

al. (1996). For those areas supporting western juniper/mountain big sagebrush and western juniper/low sagebrush woodland PVTs, successional stages identified included herbland, shrubland, and initiation, open young, young multistory, and mature woodlands with 0, <2, <5, 5–10, >10, and >15% western juniper canopy coverage, respectively (Table 1). Stand initiation, open young, and young multistory woodlands correspond to the Phase 1, Phase 2, and Phase 3 successional stages, respectively, as described by Miller et al. (2005).

Landsat Classification Procedures

The land cover PVT/structural stage map was produced through a supervised classification (Roth 2004) of a Landsat 7 ETM+ image from 2 August 2002. Image preprocessing steps included radiometric correction and georeferencing to U.S. Geological Survey orthophotos. The classification of the Landsat spectral bands was complemented by the use of topographic variables, elevation and slope, and a solar radiation variable. The solar radiation variable was derived from the 30-m digital elevation model through processing in the Solar Analyst software (HEMI Solar Analyst 1999–2000). The image was classified into successional stages of each PVT using a supervised non-parametric nearest neighbor discriminant analysis method (SAS 8.02; SAS Institute, Cary, NC). The overall accuracy of the land cover classification was 72%. Post-processing steps included smoothing of the

image through filtering in ArcInfo (ESRI, Redlands, CA) and conversion of the raster image to polygons.

A total of >740 ground control points were collected in the watersheds in 1999–2003. Approximately 50% of the points were used as training points for the classification of satellite imagery, and the remaining points were reserved for accuracy assessment. The PVT and successional stage was recorded for each point. Location of ground reference points were recorded using Magellan (Santa Clara, CA) Global Positioning System units with an estimated maximum spatial error of 15 m. Large (>2 ha) homogeneous areas were selected for the ground control to ensure that the control area was larger than the minimum mapping unit and that the effects of the GPS spatial error would be minimized. Each ground control point was in the center of homogeneous vegetation with a minimum radius of 100 m.

Succession modeling is critical in order to predict broadscale long-term landscape trends (Jensen and Bourgeron 1993, Kaufmann et al. 1994). The Tool for Exploratory Landscape Scenario Analyses (TELSA; Version 3.3), a spatially explicit, deterministic succession modeling tool with stochastic properties, was utilized to predict future landscape compositions that would result from variety of fire management strategies (Kurz et al. 2000a,b; <http://www.essa.com/downloads/telsa/index.htm>). Changes in future landscape composition given different land management strategies were modeled using TELSAs (Klenner et al.

2000). The model predicts post-fire succession from a process developed by Kessell and Fischer (1981). This approach links seral vegetation change along a multiple pathway model of successional development as initially suggested by Noble and Slatyer (1977). The length of time a given landscape unit is occupied by a particular successional stage is determined by life-span and recruitment strategy of the dominant canopy species. This approach has been successfully used to model successional change in many forested vegetation types on the Northern Rocky Mountains (Crane and Fischer 1986, Fischer and Bradley 1987, Bradley et al. 1992, Klenner et al. 2000).

Succession was modeled on an individual polygon scale as a change in successional stage within each PVT. Following the initial image classification, each patch was tessellated into irregular-shaped polygons that were approximately 1 ha in area. The initial conditions for each polygon are set utilizing information on the PVT and the current successional stage. The occurrences of human- and natural-caused disturbances are stochastically simulated at the polygon level from a probability matrix. Disturbances result in an immediate change in successional stage, with a corresponding adjustment of the successional clock. Disturbance probabilities can be modified to assess landscape changes that would be affected by various fire management strategies (Klenner et al. 2000; Kurz et al. 2000*a,b*). Probability of the size class of wildfires and prescribed fires were defined and randomly assigned to each fire. The majority of wildfires were <1 ha (90%), while the majority of prescribed fires were 100–1,000 ha in area (70%). In the modeling process, tessellated polygons were randomly selected for burning based on the predetermined fire disturbance probability function assigned to each cover type–PVT combination. If a polygon was randomly selected for burning by either wildfire or prescribed fire, the fire was randomly assigned an area based on the predetermined fire size probability function. Bare rock or cover types such as those with shallow soils that do not burn were given a zero disturbance probability and thus did not burn in this process.

Future landscape compositions of the 3 HUCs were evaluated for 50, 100, and 200 y into the future. Fire management regimes assessed for each watershed included

- 1) Current fire management (suppressed wildfire only).
- 2) Use of prescribed fire to treat a combined average of 2% of the watershed area/decade focusing on Phase 1 and Phase 2 woodland successional stages.
- 3) Use of prescribed fire to treat an annual combined average of 5% of the watershed area/decade focusing on Phase 1 and Phase 2 woodland successional stages.
- 4) Use of prescribed fire to treat an annual combined average of 7% of the watershed area/decade focusing on Phase 1 and Phase 2 woodland successional stages.

Using a Monte Carlo approach, TELSA was run

10 times for each management regime per watershed. Fire occurrence was applied at random within each PVT and phase combination for each watershed. Means and variances were then calculated from these results.

Landscape metrics for the current and predicted vegetations were calculated using FRAGSTATS 3.3 (McGarigal and Marks 1994). This software performs landscape analysis using a number of indices. Four indices were selected that reflect the different types of changes in landscape composition and pattern. The indices selected to quantify the landscape changes for this study were Simpson's evenness index (SEI), mean patch size (MPS), contagion (C), and interspersed-juxtaposition index (IJI). The SEI is a measure of community diversity that is determined by abundance of each patch type within a watershed and decreases as a patch type becomes more dominant within a watershed. The SEI was selected rather than other diversity indices such as Simpson's diversity index or the Shannon–Wiener diversity index because community richness did not change over the modeled time periods. The SEI is a more sensitive measure of changes in the proportion of landscape cover types. The MPS is an average size of all patch types within a landscape. It averages the effects of both the aggregation of patches through succession and the creation of new patch types through disturbance. Patch size can have important effects on the modification of the local environment and the selection of habitat by many species. The C is related to the distribution of each patch type within the watershed and decreases as the spatial distribution of a patch type becomes more aggregated. This metric is affected by both disturbance patterns and more constant factors such as the distribution of soil type, geological parent materials, and aspect. The IJI is a measure of the randomness of patch type adjacency. IJI decreases as patch type adjacencies (cover type edges) become less random and as a patch type is associated more frequently with other patch types. The distribution of different types of edges influences some species' habitat selection and the degree of microclimate modification by edges. Together these four metrics measured the effects of disturbance on cover type diversity, mean patch size, patch type distribution, and diversity of edge types. Significant differences between management strategies were detected with analysis of variance and Tukey's pairwise mean comparisons test ($P \leq 0.05$) (SYSTAT 10.0; SPSS, Chicago, IL).

RESULTS

Under current fire management, western juniper woodlands were predicted to continue maturing as has been noted in previous field studies (Burkhardt and Tisdale 1969, Miller and Wigand 1994, Miller et al. 2005). Mature western juniper stages were predicted to increase about 30% in area over the next 100 y on both mountain big sagebrush and low sagebrush sites (Table 2). Phase 3 woodlands were predicted to in-

Table 2. Total predicted area (ha) in mountain big and low sagebrush steppe and Phase 1 juniper woodland cover types for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes (mean \pm SD).

Watershed	Management regime	Time period			
		Present	50 y	100 y	200 y
Currant Creek ^a	Currant fire management	1,087	740 \pm 19 a ^b	714 \pm 17 a	517 \pm 11 a
	Prescribed fire (2%/decade)	1,087	900 \pm 106 b	858 \pm 109 b	636 \pm 81 a
	Prescribed fire (5%/decade)	1,087	972 \pm 185 b	1,108 \pm 173 c	896 \pm 209 b
Red Canyon Creek	Current fire management	1,908	445 \pm 8 a	426 \pm 9 a	305 \pm 5 a
	Prescribed fire (2%/decade)	1,908	600 \pm 12 b	504 \pm 23 b	343 \pm 15 b
	Prescribed fire (5%/decade)	1,908	1,188 \pm 23 c	936 \pm 27 c	900 \pm 29 c
Smith Creek	Prescribed fire (7%/decade)	1,908	1,539 \pm 28 d	1,194 \pm 38 d	1,321 \pm 178 d
	Current fire management	1,247	459 \pm 7 a	395 \pm 12 a	343 \pm 13 a
	Prescribed fire (2%/decade)	1,247	647 \pm 152 ab	470 \pm 110 a	205 \pm 47 a
	Prescribed fire (5%/decade)	1,247	832 \pm 387 b	1,106 \pm 367 b	837 \pm 275 b
	Prescribed fire (7%/decade)	1,247	1,700 \pm 312 c	1,848 \pm 383 c	1,396 \pm 255 c

^a The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.

^b Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

crease 200% and 120%, respectively, for mountain big and low sagebrush site over the next 100 y. The increased area of later successional stages was derived from earlier successional stages, such as sagebrush steppe, and Phase 1 and Phase 2 woodlands as they continue to advance in successional age. This process resulted in a reduction of SEI as the watersheds become increasingly dominated by later successional stages (Table 3). The MPS was predicted to decrease (Table 4) and C was predicted to increase (Table 5). In 100 y, sagebrush-dominated stages were predicted to nearly disappear from the watersheds. The decline of mountain big sagebrush communities was predicted to be greater than that of the low sagebrush-dominated communities because of faster rates of succession associated with that PVT.

Fire is frequently suggested as a potential management tool to counteract the influence of woodland encroachment (Wright and Bailey 1982, Miller et al. 2005). We compared the efficacy of using current management to three different levels of prescribed fire to maintain sagebrush-dominated vegetation. We simulated burning approximately 2, 5, and 7% of each watershed per decade, focusing burning the early stages of woodland succession (Phases 1 and 2). The 2% lev-

el was found to be inadequate to maintain the current amount of sagebrush steppe on the landscape (Table 2). Woodland continued to develop at the expense of sagebrush steppe. The 5% level of fire was adequate in maintaining low sagebrush steppe for all watersheds within the next 100 y. The 5% level of fire was also adequate in maintaining mountain big sagebrush steppe for the Smith Creek and Currant Creek watersheds. However, for Red Canyon Creek, which currently has 21% of mountain big sagebrush steppe, more than 7% per decade will be required to be burned. The 7% level resulted in a reduction in mountain big sagebrush steppe in the Currant Creek watershed. This decrease occurred because, at this level, the non-woodland areas were being burned so frequently that sagebrush steppe did not have adequate time to develop. Thus, grassland vegetation increased in the watersheds. In the long term (>200 y), Phase 3 woodlands continued to increase under all management regimes. This increase was due to the inability to burn all Phase 2 woodland areas. These areas advanced through succession into Phase 3 woodlands that were not subject to the prescribed fire regimes.

The changes in SEI and C through time for the four regimes were fairly consistent, although the mag-

Table 3. Predicted Simpson's evenness index for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes for three future time periods (mean \pm SD).

Watershed	Management regime	Time period			
		Present	50 y	100 y	200 y
Currant Creek ^a	Currant fire management	0.927	0.921 \pm <0.001 a ^b	0.912 \pm 0.001 a	0.899 \pm 0.001 a
	Prescribed fire (2%/decade)	0.927	0.921 \pm 0.001 ab	0.914 \pm 0.002 b	0.894 \pm 0.002 b
	Prescribed fire (5%/decade)	0.927	0.919 \pm 0.002 ac	0.919 \pm 0.002 c	0.894 \pm 0.004 b
Red Canyon Creek	Current fire management	0.918	0.900 \pm <0.001 a	0.827 \pm 0.001 a	0.829 \pm 0.001 a
	Prescribed fire (2%/decade)	0.918	0.912 \pm 0.001 b	0.853 \pm 0.002 b	0.830 \pm 0.001 a
	Prescribed fire (5%/decade)	0.918	0.931 \pm 0.001 c	0.912 \pm 0.001 c	0.875 \pm 0.001 b
Smith Creek	Prescribed fire (7%/decade)	0.918	0.928 \pm 0.001 d	0.925 \pm 0.001 d	0.895 \pm 0.001 c
	Current fire management	0.957	0.910 \pm 0.001 a	0.880 \pm 0.003 a	0.881 \pm 0.001 a
	Prescribed fire (2%/decade)	0.957	0.922 \pm 0.001 b	0.889 \pm 0.010 a	0.873 \pm 0.005 a
	Prescribed fire (5%/decade)	0.957	0.936 \pm 0.001 c	0.926 \pm 0.032 b	0.907 \pm 0.013 b
	Prescribed fire (7%/decade)	0.957	0.938 \pm 0.001 c	0.938 \pm 0.023 c	0.919 \pm 0.008 c

^a The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.

^b Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

Table 4. Predicted mean patch size (ha) for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes for three future time periods (mean ± SD).

Watershed	Management regime	Time period			
		Present	50 y	100 y	200 y
Currant Creek ^a	Currant fire management	12.9	10.7 ± 0.1 a ^b	9.9 ± 0.2 a	9.4 ± 0.1 a
	Prescribed fire (2%/decade)	12.9	9.7 ± 0.6 b	9.2 ± 0.4 b	9.7 ± 0.6 ab
	Prescribed fire (5%/decade)	12.9	10.1 ± 0.4 b	9.1 ± 0.3 b	9.2 ± 0.5 ac
Red Canyon Creek	Current fire management	12.4	10.7 ± 0.1 a	10.7 ± 0.1 a	9.2 ± 0.1 a
	Prescribed fire (2%/decade)	12.4	8.9 ± 0.2 b	9.0 ± 0.1 b	9.1 ± 0.2 a
	Prescribed fire (5%/decade)	12.4	7.3 ± 0.1 c	6.2 ± 0.1 c	6.7 ± 0.2 b
Smith Creek	Prescribed fire (7%/decade)	12.4	7.7 ± 0.1 d	6.1 ± 0.1 c	5.8 ± 0.1 c
	Current fire management	10.6	10.7 ± 0.1 a	10.8 ± 0.3 a	10.4 ± 0.3 a
	Prescribed fire (2%/decade)	10.6	9.5 ± 1.2 b	10.2 ± 1.0 a	12.4 ± 0.8 b
	Prescribed fire (5%/decade)	10.6	7.9 ± 1.1 c	7.4 ± 1.0 b	8.1 ± 1.4 c
	Prescribed fire (7%/decade)	10.6	8.3 ± 0.9 c	7.0 ± 0.5 b	6.9 ± 0.7 d

^a The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.

^b Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

nitude of change varied by watershed (Tables 3, 5). The SEI decreased steadily though 100 y for all regimes. The decrease was greatest for the current management and 2% regimes. The lowest amount of decrease was predicted for the 7% regime. However, the low amount of Phase 1 and Phase 2 woodlands currently present in Currant Creek did not permit burning 7%. The C increased for the current management and 2% regimes up through 100 y into the future. It did not change greatly for any watershed under the 5% and 7% regimes.

The IJI was the most inconsistent pattern metric analyzed (Table 6). It increased for all fire management regimes at 50 y in Red Canyon Creek but remained relatively unchanged in Currant Creek and Smith Creek during this time period. At 100 y in the future IJI increased in Red Canyon, decreased in Currant Creek, and remained constant in Smith Creek under the 5% regime.

The MPS was predicted to decrease over the 100-y period for all watersheds with increasing fire occurrence (Table 4). The lowest decrease was predicted for Smith Creek. Smith Creek had the smallest initial patch size (10.5 ha) that may have influenced this effect.

DISCUSSION AND MANAGEMENT IMPLICATIONS

Results of this study suggest that encroachment of western juniper is likely to continue and will have significant effects on diversity and vegetation pattern of these watersheds. Under current fire management policy, SEI and IJI will continue to decline and C will continue to increase as juniper woodland becomes more dominant. Both mountain big sagebrush and low sagebrush steppe will decline to low levels as encroachment of juniper onto these sites continues. Rapid successional rates associated with more productive mountain big sagebrush PVT will result in <100 ha of this habitat available in each watershed within 100 y. Increased dominance of mature juniper woodland will influence future fire regimes, watershed characteristics and succession patterns, and increase the habitat for woodland species, particularly primary and secondary cavity-nesting animals. Juniper dominance will also have major negative implications for sagebrush steppe-dependent species such as the sage thrasher (*Oreoscoptes montanus*), sage sparrow (*Amphispiza belli*), and Brewer’s sparrow (*Spizella breweri*).

Table 5. Predicted contagion index for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes for three future time periods (mean ± SD).

Watershed	Management regime	Time period			
		Present	50 y	100 y	200 y
Currant Creek ^a	Currant fire management	49.0	48.7 ± 0.1 a ^b	50.5 ± 0.2 a	51.8 ± 0.2 a
	Prescribed fire (2%/decade)	49.0	48.4 ± 0.4 a	49.9 ± 0.6 b	52.6 ± 0.6 b
	Prescribed fire (5%/decade)	49.0	49.1 ± 0.7 a	48.7 ± 0.5 c	52.4 ± 1.0 b
Red Canyon Creek	Current fire management	49.4	50.2 ± 0.1 a	55.8 ± 0.1 a	55.6 ± 0.2 a
	Prescribed fire (2%/decade)	49.4	47.6 ± 0.2 b	52.5 ± 0.2 b	55.5 ± 0.1 a
	Prescribed fire (5%/decade)	49.4	44.9 ± 0.2 c	45.7 ± 0.1 c	49.3 ± 0.2 b
	Prescribed fire (7%/decade)	49.4	46.4 ± 0.1 d	44.2 ± 0.4 d	47.0 ± 0.4 c
Smith Creek	Current fire management	44.9	49.5 ± 0.1 a	52.9 ± 0.2 a	53.4 ± 0.3 a
	Prescribed fire (2%/decade)	44.9	47.6 ± 1.8 ac	51.9 ± 1.6 ab	55.3 ± 0.9 a
	Prescribed fire (5%/decade)	44.9	45.5 ± 2.4 bc	46.5 ± 2.1 bc	49.2 ± 2.5 b
	Prescribed fire (7%/decade)	44.9	46.0 ± 1.8 bc	45.1 ± 1.0 bc	47.4 ± 1.7 b

^a The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.

^b Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

Table 6. Predicted interspersed–juxtaposition index for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes for three future time periods (mean \pm SD).

Watershed	Management regime	Time period			
		Present	50 y	100 y	200 y
Currant Creek ^a	Currant fire management	78.9	80.2 \pm 0.3 a ^b	77.0 \pm 0.3 a	75.6 \pm 0.3 a
	Prescribed fire (2%/decade)	78.9	80.1 \pm 1.0 a	77.0 \pm 1.1 a	74.2 \pm 1.5 b
	Prescribed fire (5%/decade)	78.9	79.2 \pm 1.3 a	79.5 \pm 1.0 b	74.4 \pm 1.4 b
Red Canyon Creek	Current fire management	75.5	76.9 \pm 0.3 a	73.4 \pm 0.3 a	75.1 \pm 0.4 a
	Prescribed fire (2%/decade)	75.5	78.9 \pm 0.2 b	75.5 \pm 0.5 b	74.9 \pm 0.4 a
	Prescribed fire (5%/decade)	75.5	79.1 \pm 0.3 b	81.2 \pm 0.2 c	79.0 \pm 0.3 b
Smith Creek	Prescribed fire (7%/decade)	75.5	77.6 \pm 0.4 c	83.1 \pm 0.8 d	79.7 \pm 0.8 c
	Current fire management	78.0	77.0 \pm 0.2 a	72.6 \pm 0.3 a	72.0 \pm 0.5 a
	Prescribed fire (2%/decade)	78.0	77.4 \pm 2.2 a	72.0 \pm 2.4 a	67.9 \pm 1.3 b
	Prescribed fire (5%/decade)	78.0	77.5 \pm 2.1 a	76.8 \pm 1.0 b	74.9 \pm 2.6 c
	Prescribed fire (7%/decade)	78.0	77.3 \pm 1.8 a	78.2 \pm 2.1 b	75.7 \pm 2.7 c

^a The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.

^b Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

In the three watersheds studied, prescribed fire could be implemented to alleviate this trend. Prescribed burning at least 5% of each watershed per decade is necessary to maintain the current level of low sagebrush steppe currently present. The amount of fire required to maintain mountain big sagebrush steppe is more variable among watersheds and dependent upon the proportion currently present. Generally application of 5–7% of the area per decade was adequate. However, in Red Canyon Creek, the watershed with the greatest proportion of mountain big sagebrush steppe, prescribed fire at the 7% level was not able to maintain current conditions.

The modeled prescribed fire was focused on the Phase 1 and Phase 2 woodland cover types. It was evident that this approach would not sustain either low or mountain big sagebrush steppe over the long term (>200 y). With this approach, there was a slow but continuous loss to Phase 3 communities that were then precluded from prescribed fire activity. Phase 3 woodlands were modeled with a low prescribed fire probability because they only burn under the most severe fire conditions. The level of prescribed fire in Phase 1 and Phase 2 community types only slowed the rate of decline. The solution appears to be either increasing the area of Phase 2 woodlands burned per decade or treatment of Phase 3 woodlands so they can be returned to earlier sagebrush-steppe portions of the successional sequence. However, Phase 3 communities are more difficult to prescribe burn due to low fine fuel loading and require greater amounts of time to recover (Miller et al. 2000, 2005). Other land treatments, such as mechanical removal of juniper, may have to be considered for Phase 3 woodland sites.

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FIRE IN OAK WOODLANDS: A GENERAL LAND OFFICE PERSPECTIVE

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ABSTRACT

The General Land Office records of witness trees provide the reference condition for arboreal vegetation, reflect conditions circa 1815–1845, and represent the potential historic vegetation that existed prior to the extensive disturbances caused by European settlement.

The data were compiled from surveyor's notes and were linked as a layer to the ArcView geographic information system (GIS) layer for the study area. Nine land types, based on topographic position, slope, and aspect are derived from the GIS. Twenty-eight tree species were identified by common name. Species found in all land types include white oak, black gum, hickory, and red oak. Pines and black oaks are found in all land types but the floodplain. Pines were present in the toe slope adjacent to the floodplain.

Four basic indices were produced and are comparable to modern botanical analyses:

Relative density: number of times a species occurs out of the total N observations;

Basal area: sum of all diameters–species–landtype–landtype association;

Relative dominance: total basal area of each species divided by basal area of all species;

Importance value: add the relative density of each species to the relative dominance.

In the floodplain, three species dominate: white oak (31.6%), black gum (18.4%), and elm (13.2%). The toe slope is adjacent to the floodplain and contains five dominant species: white oak (33.3%), black oak (13.3%), pine (11.1%), black gum (7.7%), and beech (7.7%).

The north aspect is divided into three parts: lower, middle, and upper slope. The lower slope is dominated by five species: white oak (43.7%), beech (25%), red oak (6.2%), black oak (6.2%), and pine (6.2%). The middle slope is dominated by five species: white oak (43.8%), black oak (16.9%), pine (10%), hickory (5.8%), and black gum (5.4%). The upper slope is dominated by four species: white oak (45.7%), black oak (14.9%), black gum (10.2%), and pine (7.1%).

The south aspect is also divided into lower, middle, and upper slopes. The lower slope is dominated by three species: white oak (29.4%), black oak (23.5%), and pine (20.5%). The middle slope is dominated by five species: white oak (29.3%), pine (28.5%), black oak (17.6%), red oak (4.7%), and black gum (4.4%). The upper slope is dominated by five species: white oak (34.1%), pine (21.5%), black oak (20.5%), red oak (6.8%), and hickory (6.3%).

The uplands are characterized by rolling to broken flat ridgetops. These areas are dominated by three species: white oak (43%), black oak (22.9%), and pine (13.8%).

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EFFECTS OF FIRE ON VEGETATION DYNAMICS IN TALLGRASS PRAIRIE: 30 YEARS OF RESEARCH AT THE KONZA PRAIRIE BIOLOGICAL STATION

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ABSTRACT

Fire is a key factor shaping the structure and dynamics of tallgrass prairie ecosystems. Since 1972, studies at the Konza Prairie Biological Station (KPBS) have been examining the effects of fire on plant and animal populations and communities, and on ecosystem-level processes. The 3,487-ha preserve is divided into 55 watershed units (average size = 60 ha), each subjected to a specific combination of prescribed fire frequency (burned in the spring at 1-, 2-, 4-, 10-, or 20-y intervals) or fire season (burned in February, April, July, or November), and grazing treatment (grazed by bison [*Bison bison*], cattle, or ungrazed). These experimental treatments, replicated at the watershed level, allow large-scale studies of the effects of varying fire regimes, and their interaction with grazing effects, on tallgrass prairie.

These long-term studies at KPBS have shown that frequent fire in tallgrass prairie (e.g., annual or biannual spring burning) increases the competitive dominance and relative abundance of warm-season perennial C₄ grasses and decreases the relative abundance of subdominant C₃ graminoids and forbs, resulting in a decrease in plant species richness and diversity. Frequent fire also significantly reduces the cover and frequency of woody plant species, and in the absence of fire, complete displacement of native prairie by woody vegetation can occur in <4 decades. Long-term study at KPBS has also shown that frequent fire reduces the frequency of exotic grass and forb species in tallgrass prairie plant communities. The reduction in floristic diversity due to frequent fire is offset by grazing, which increases plant species diversity. Plant species richness is not significantly affected by season of burn. Plant species respond differentially to season of fire, although there is a general increase in the relative abundance of perennial forbs under frequent dormant-season (autumn or winter) fire.

These vegetation responses are driven by fire effects on nutrient cycling processes and on shifts in the relative limitation of key plant resources (light, water, and nutrients). Many plant population and community responses to fire in tallgrass prairie are mediated by fire effects on mutualistic symbiotic associations between plants and arbuscular mycorrhizal fungi.

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FIRE-INDUCED CHANGES IN SOIL NITROGEN AND CARBON DYNAMICS IN TALLGRASS PRAIRIE

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ABSTRACT

Periodic fires are essential for the development and persistence of relatively mesic grasslands, including the tallgrass prairies of the U.S. Central Plains. Today, fire is widely used as a management tool to limit encroachment by woody plants and to maintain the productivity of the dominant C_4 grasses in these grasslands. Fires at different frequencies affect many aspects of tallgrass prairie ecosystem structure and functioning, including plant species composition, plant productivity, plant tissue chemistry, and the soil microclimate, all of which can affect soil processes and soil biota. Long-term studies at the Konza Prairie Long Term Ecological Research site have been evaluating belowground responses to fire at different frequencies (i.e., annual spring fires versus infrequent fires or long-term fire exclusion). Here we summarize some of the major findings of these studies with respect to belowground nitrogen (N) and carbon (C) cycling. Fire results in the volatilization of N from aboveground detritus in tallgrass prairie, while subsequent changes in the light and energy environment generally increase both above- and belowground plant productivity. Increased root productivity and increased N use efficiency by the dominant grasses leads to greater inputs of plant material to the soil and a wider C/N ratio of those inputs, both of which influence soil N and C flux. Soils under frequently burned prairie exhibit lower N availability, evidenced by reduced concentrations of inorganic N and 33–66% lower rates of soil net N mineralization, relative to unburned prairie. Soils of frequently burned prairie also exhibit less spatial heterogeneity in available soil N, with potential consequences for plant community dynamics. In contrast to the effects of fire on N mineralization, total soil CO_2 efflux is increased by up to 55% in frequently burned, relative to unburned, prairie. Root productivity and biomass are also higher in frequently burned prairie. Comparisons of soil C and N dynamics in annually burned and infrequently burned prairie with similar soil microclimates suggest that abiotic factors alone (e.g., altered soil temperature and moisture) cannot account for these changes and that biotic factors are key to explaining belowground responses to fire in tallgrass prairie.

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FIRST-YEAR RESPONSE TO SUMMER FIRE AND POST-FIRE GRAZING EFFECTS IN NORTHERN MIXED PRAIRIE

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ABSTRACT

Summer wildfire is a common occurrence in the northern mixed prairie. Two-year deferment from grazing is generally recommended following wildfire. However, little research has been conducted to determine whether deferment is necessary for vegetative recovery. Research objectives were to determine if summer fire and post-fire grazing at different utilization levels would affect standing crop, current-year biomass, and species composition. Five treatments were applied to twenty 0.75-ha plots near Miles City, Montana, with four replications of each treatment. Treatments were no burn + no graze, burn + 0% utilization, burn + 17% utilization, burn + 34% utilization, and burn + 50% utilization. Fire was applied 29 August 2003, and grazing treatments were applied late June through early July 2004. Current-year biomass was similar between burned and unburned plots for herbaceous components. Total standing crop and current-year biomass were each similar among utilization levels. Fire decreased threadleaf sedge (*Carex filifolia*) and needle-and-thread (*Heterostipa comata*) by 106 and 136 kg ha⁻¹, respectively. Annual bromes (*Bromus* spp.) and fringed sage (*Artemisia frigida*) were minor components, but were reduced by fire as well. Western wheatgrass (*Pascopyrum smithii*), other cool-season perennial grasses, and forbs were unaffected by fire. Warm-season perennial grasses, predominantly blue grama (*Bouteloua gracilis*), were reduced 156 kg ha⁻¹ by fire and decreased linearly with increasing utilization. Early results indicate that summer fire and grazing may alter species composition but do not reduce current-year biomass across species during a drought year.

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EFFECTS OF SEASONAL FIRES ON THE TEMPORAL STABILITY OF HERBACEOUS PRODUCTION IN A MESQUITE-ENCROACHED GRASSLAND

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ABSTRACT

In Great Plains ecosystems, fire has long been used to suppress shrub encroachment and augment herbaceous production. Accordingly, the effects of fire on vegetative physiognomy and herbaceous production have received much attention. However, the effect of fire on the temporal stability of herbaceous production has received relatively little attention. Here we evaluate the results of repeated seasonal fires on the post-treatment temporal stability of herbaceous production within honey mesquite (*Prosopis glandulosa*)-encroached grasslands. We define temporal stability as the coefficient of variation for herbaceous biomass ($CV = 100 \times 1 \text{ SD}/\text{mean}$) within each treatment plot across years. In one analysis, we examine the effects of repeated winter fires or repeated summer fires in combination with either simulated grazing or no grazing on the temporal stability of an ecologically and economically important C_4 midgrass. In a second analysis, we examine the effects of repeated summer fires, repeated winter fires, and alternating winter–summer fires on the temporal stability of a complete herbaceous community exclosed from livestock grazing. Preliminary results indicate summer fire decreases temporal stability, while winter fire and grazing have little effect. Post-fire mesquite regrowth is highest in winter burn treatments and likely lowers herbaceous production but increases temporal stability. Potential effects of seasonal fires, or the resultant alteration in shrub physiognomy, on the temporal stability of herbaceous production are important information for land managers, especially in arid and semiarid environments where precipitation varies greatly between years. Long-term, post-fire biomass data are needed to determine how fires affect processes such as temporal stability and drought resilience.

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EFFECTS OF SUMMER FIRES ON WOODY, SUCCULENT, AND GRAMINOID VEGETATION IN SOUTHERN MIXED-PRAIRIE ECOSYSTEMS: A REVIEW

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ABSTRACT

Interest in the use of summer fires to restore southern Great Plains prairie ecosystems infested with woody plants and cactus is increasing, but information regarding effects on target and nontarget plant species is needed before this practice can be recommended. We review effects of summer and winter fires on mortality and growth of honey mesquite (*Prosopis glandulosa*), prickly pear (*Opuntia phaeacantha*), and grasses common to the southern mixed prairie, including C₄ midgrass sideoats grama (*Bouteloua curtipendula*), C₄ shortgrass buffalograss (*Buchloe dactyloides*), and C₃ midgrass Texas wintergrass (*Nassella leucotricha*). Summer fires were more effective than winter fires at top-killing mature mesquite, but plant mortality was <4% in all fire treatments. Prickly pear plant mortality was much greater following summer than winter fires. Sideoats grama, buffalograss, and Texas wintergrass were all tolerant of summer fire, although post-fire recovery rate was slower in sideoats grama than in the other species. Clipping once each spring reduced sideoats grama and Texas wintergrass standing crop in all treatments when measured 1 y after clipping, but effects of clipping + summer fire were not negatively additive. The only time fire negatively affected Texas wintergrass was when spring clipping followed winter fire. In contrast, buffalograss standing crop was not reduced by clipping or clipping + summer fire at 1 y after treatment. Results suggest that summer fire will not eradicate mesquite and that frequent fires either in summer or winter are necessary to maintain suppression. Grasses studied thus far have demonstrated a tolerance to fire in either season.

keywords: cactus, grassland, herbaceous production, mesquite, prescribed fire, southern Great Plains, species composition, summer fire, woody plants.

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INTRODUCTION

Prescribed fires in the southern Great Plains prairie have historically been conducted during the dormant season (January–March) because they are safer and more manageable than growing-season, or “summer,” fires (Wright and Bailey 1982, Scifres and Hamilton 1993). However, there has been increasing interest in the use of summer-season fires because of their greater ability to suppress or even kill noxious woody plants and cacti species (Ansley and Jacoby 1998, Taylor 2001, Ansley and Taylor 2004). While the potential controlling effects of summer fires on target noxious species holds promise, there is concern that summer fires may damage economically and/or ecologically important nontarget herbaceous species (Bailey 1988) or drastically reduce grass production (Engle and Bultsma 1984, Engle and Bidwell 2001). Very few data are available that document responses of target or nontarget species in replicated studies that compare summer fires, winter fires, and an unburned control. Our purpose is to provide a historical context for and to summarize the effects of summer fires on Great Plains vegetation, with particular emphasis on plant species in the southern Great Plains.

Historically, Great Plains vegetation was most probably maintained as grassland because of the frequent occurrence of fires (Archer 1989, Collins and Wallace

1990, Van Auken 2000). This conclusion is not new, and it is surprising how many early observers realized that the vast grasslands of the Great Plains were maintained by fire and would experience woody plant encroachment if fire was removed from these ecosystems. As early as 160 y ago, Josiah Gregg, probably writing about western Oklahoma (Stewart et al. 2002), commented on how fires (called “conflagrations”) maintained the southern prairie grasslands (Gregg 1844:202):

It is unquestionably the prairie conflagrations that keep down the woody growth upon most of the western uplands. The occasional skirts and fringes which have escaped their rage have been protected by the streams they border. Yet, may not the time come when these vast plains will be covered with timber? . . . Indeed there are parts of the southwest now thickly set with trees of good size that within the remembrance of the oldest inhabitants were as naked as the prairie plains and the appearance of the timber in many other sections indicates that it has grown up within less than a century. In fact, we are now witnessing the encroachment of the timber upon the prairie wherever the devastating conflagrations have ceased their ravages.

Bray (1901:209) in western Texas, even acknowledging then that this was not a new idea, stated:

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Apparently under the open prairie regime the equilibrium was maintained by more or less regular recurrence of prairie fires. This, of course, is by no means a new idea, but the strength of it lies in the fact that the grass vegetation was tolerant of fires and the woody vegetation was not. It was only after weakening the grass floor by heavy pasturing and ceasing to ward off the encroaching species by fire that the latter invaded the grass lands.

Cook (1908:1–16) wrote about fire and woody encroachment in South Texas prairies:

That such fires were evidently the cause of the former treeless condition of the southwestern prairies is also shown by the fact that trees are found in all situations which afford protection against fires.

Foster (1917:442–445) wrote, concerning central Texas:

The causes which have resulted in the spread of timbered areas are traceable directly to the interference of man. Before the white man established his ranch home in these hills the Indians burned over the country repeatedly and thus prevented an extension of forest areas. With the settlement of the country grazing became the only important industry. . . .

. . . Overgrazing has greatly reduced the density of grass vegetation. . . . Almost unquestionably the spread of timbered areas received its impetus with the gradual disappearance of grassland fires and has been hastened by the reduction of the grass cover itself.

Recent review articles by Daubenmire (1968), Axelrod (1985), Anderson (1990), Bock and Bock (1995), Van Auken (2000) and Briggs et al. (2005) have all reaffirmed that fires were common and necessary to maintain Great Plains grasslands. Sauer (1950) and Bock and Bock (1995) concluded that there is no climatic condition that favors grassland over woodland. In other words, there is no combination of precipitation and temperature that is sufficient to allow grassland to replace desert, but that is insufficient for woodland to replace grassland. Thus, without fire, the soils and climate that support grasslands could, in most cases, support a shrubland or woodland as well.

Frost (1998) suggested the fire return interval in most of the Great Plains was <6 y, and in certain regions, <3 y. It is thought that most fires were caused by lightning strikes of dry vegetation. However, fires in some areas were ignited by Native Americans, either to assist hunting large game by surrounding them with a ring of fire, or kill small mammals and reptiles for food, or use as a means to stimulate lush herbaceous growth the following year to either attract game, such as bison (*Bison bison*), or to increase yields of seeds and berries (Axelrod 1985, Stewart et al. 2002). In some instances, they may have used fire to clear areas of brush or fires occurred accidentally (Foster 1917, Stewart et al. 2002).

Because lightning strikes are more frequent during summer thunderstorms, many fire ecologists believe that summer wildfires were more common than dormant-season (i.e., winter or spring) fires (Higgins 1986, Ewing and Engle 1988, Taylor 2001). The timing of the ignition of fires by Native Americans on southern Great Plains grasslands is largely unknown. Higgins (1986) indicated that, in the northern mixed-grass prairie of the Dakotas, most fires were ignited by Native Americans during fall (October) or early spring (April). Stewart et al. (2002) quoted several early explorers near St. Louis who witnessed late autumn burning by Native Americans. Jared Smith, referring to central Texas, wrote in 1899 that, “Weeds and brush were kept in check by the fires set by the Indians in early spring to improve pasturages. In this manner the encroaching of thorny shrubs, cactus and mesquite was prevented. . . .” (Smith 1899:7–8). While traveling in Oklahoma in October 1832, Washington Irving reported that he observed daily “haziness” from fires being ignited by Native Americans (Stewart et al. 2002:150). In contrast, Frost (1998) suggested that burning by Native Americans in the southern mixed-prairie regions of Oklahoma and North Texas coincided with lightning strike peaks in midsummer.

It is our opinion that it was more likely for Native Americans in the southern prairie to conduct dormant-season rather than summer fires, especially if the goal was to provide green growth in spring to attract bison and other game. Thus, if we make a very simplistic generalization and imagine that most lightning-strike fires occurred in summer and most Native American-ignited fires occurred during the dormant months (late fall, winter, or early spring), it is conceivable that fire could potentially have occurred most months of the year in the southern prairie. Thus, the long-term result prior to European settlement may have been a continually changing mosaic of summer-burned, dormant season-burned, and unburned patches of widely ranging and often overlapping sizes.

For this paper, we assumed that summer fires were common to the southern Great Plains prior to European settlement (Engle and Bidwell 2001). We review some of the research that has been conducted regarding responses of certain plant species to summer fires. In particular, we focus on our research in North Texas that has emphasized direct contrasts between summer fire and dormant-season (i.e., “winter”) fire effects within the same experimental framework on the woody legume honey mesquite (*Prosopis glandulosa*), brown-spined prickly pear cactus (*Opuntia phaeacantha*), the C₄ mid-grass sideoats grama (*Bouteloua curtipendula*), the C₄ shortgrass buffalograss (*Buchloe dactyloides*), and the C₃ midgrass Texas wintergrass (*Nassella leucotricha*). Research from other locations is also reviewed where it most strongly relates to our studies, but this is by no means a comprehensive review.

MESQUITE RESPONSE TO SUMMER FIRES

Most of the early research on fire effects on honey mesquite focused on effects of late-winter or early

spring fires (Wright and Bailey 1982). This research provided little evidence that single winter fires will kill adult mesquite trees. Wright et al. (1976) and Britton and Wright (1971) found moderate adult honey mesquite mortality (50% and 32%, respectively) following single late-winter (March) fires in West Texas. However, the reason for even this level of mortality was that the mesquite had been sprayed with a top-killing herbicide (2,4,5-T) 2–5 y prior to burning, and when fire was applied, the standing dead stems from the herbicide treatment ignited and burned into live root crowns, killing the buds that would have developed basal sprouts. Because these fires occurred so soon after a herbicide treatment, we view these results as responses to a combined herbicide–fire treatment, rather than to effects of fire alone.

In contrast, Ansley and Jacoby (1998) found that no adult mesquite trees were killed following single high-intensity winter fires (average 2,435 kW/m intensity, 2.4 m flame height, 2,800 kg/ha fine fuel) in North Texas. In this study, mesquite had also been sprayed with top-killing herbicides, but fire treatments were not implemented until 11–26 y after the herbicide treatments. Because of this delay, most of the standing dead stems from the herbicide treatments had decayed and disappeared. This may explain why root-kill response to fire was so low. On another site in North Texas, Ansley and Castellano (2006) found that high-intensity winter fires (flame heights 2–5 m, fine fuel 3,150 kg/ha) killed only 3% of 2- to 3-m-tall honey mesquite.

The literature indicates that honey mesquite is not susceptible to summer fires, with respect to plant mortality, although few studies have quantified this. Less than 3% of adult mesquite trees were completely killed by intense summer fires (average 4,042 kW/m intensity), even after 2 summer fires in 3 y or consecutive summer fires and under moderate herbaceous fine fuel loads (2,800 kg/ha) (Ansley and Jacoby 1998). Recent studies in southern New Mexico found similar results for 10-y-old honey mesquite (height 0.8 m) (Drewa et al. 2001) or 0.5-m-tall honey mesquite (Drewa 2003). In South Texas, late-summer fires caused 10% honey mesquite mortality (Box et al. 1967). However, this evaluation was conducted <1 y post-fire and possibly overestimated mortalities because basal sprouting following top-kill is often delayed the first growing season post-fire.

Aboveground mortality (i.e., top-kill) of adult multistemmed honey mesquite has been found to be greater following summer fires than winter fires, but this may depend on composition of the herbaceous fine fuel vegetation. On a site that had a mixture of C₃ and C₄ grasses, top-kill was greater following summer (93%) than winter (33%) fire (Ansley and Jacoby 1998). However, on a site dominated by C₄ grasses, there was little difference in top-kill between summer (86%) and winter (70%) fires. The C₃ grasses on the mixed site (mainly Texas wintergrass) were somewhat green during winter fires, and this lowered fire intensity and reduced fire impact on mesquite canopies.

Research in Arizona during the last 60 y has investigated the effects of annual or biennial summer fires

(mostly in June) on velvet mesquite (*Prosopis glandulosa* var. *velutina*). Humphrey (1949) found that summer fires caused 50% velvet mesquite mortality. Glendening and Paulson (1955) found only 15% mortality in adult velvet mesquite, but 52% mortality in mesquite seedlings following summer fire. Cable (1965) reported that a June fire killed 25% of mesquite on a high herbaceous fuel area (5,000 kg/ha), but only 8% on a lower fuel area (2,400 kg/ha). However, this evaluation was conducted at 1 y post-fire and possibly overestimated mortalities. Cable (1967) reported that a June fire reduced velvet mesquite density by 26% at 3 y post-fire. Reynolds and Bohning (1956) found 28% velvet mesquite mortality 2 growing seasons after June fires.

In studies that directly contrasted effects of summer and winter fires, Glendening and Paulson (1955) found that summer fires in June killed 29% of velvet mesquite, while winter fires killed only 4%. Blydenstein (1957) found that summer fires killed 5% of velvet mesquite, while winter fires yielded only 1% mortality. Thus, in these studies in Arizona, mortality of mature velvet mesquite from summer fires ranged from 5 to 50%, and averaged 23%.

In summary, these results indicate that one or two fire events, either in winter or summer, will not kill many adult honey mesquite trees. On sites with a high percentage of C₃ grasses, summer fire has a greater potential than winter fire for top-killing mesquite. Summer fire has caused greater mortality on adult velvet mesquite in Arizona than on honey mesquite in Texas and New Mexico.

It remains unknown as to whether or not a historical pattern of frequent fire was successful in killing the majority of honey mesquite plants that encroached into the southern Great Plains, although this is widely assumed (Gregg 1844, Van Auken 2000, Stewart et al. 2002). Wright et al. (1976) found that nearly 100% of honey mesquite seedlings that were 1.5 y old or younger were killed by fire, but mortality ranged from 20 to 72% in 2.5-y-old seedlings, depending on fire temperature, and was only 8% in 3-y-old seedlings. Thus, mesquite developmental resistance to fire may occur more rapidly than what is believed to be the historical fire regime for the southern Great Plains (Frost 1998).

PRICKLY PEAR RESPONSE TO SEASONAL FIRES

Little research has been published that has contrasted effects of summer and winter fires on prickly pear cactus. In a recent study, we (Ansley and Castellano 2007a) compared brown-spined prickly pear response to summer fires (7,446 kW/m intensity), high-intensity winter fires (4,314 kW/m), low-intensity winter fires (791 kW/m), and unburned controls. At 3 y post-treatment, mortality of individual prickly pear mottes (size ranging from 10 to 500 pads per motte) was 86% in summer fire and <16% in all other treatments. Similarly, Taylor (2001), in the Edwards Plateau of south-central Texas, found that summer and winter fires reduced prickly pear density by 97% and

47%, respectively. Thus, while winter fires may have some effect on reducing prickly pear, summer fires are clearly superior in reducing density of this species when it has become dominant.

These results differ from the findings of Bunting et al. (1980), who demonstrated that winter fires effectively killed brown-spined prickly pear in West Texas. For reasons explained earlier with honey mesquite, the region where the study of Bunting et al. (1980) occurred is dominated by C_4 grasses instead of C_3 grasses, and one might expect fewer differences in responses to summer and winter fires. However, effects of summer and winter fires were not directly compared in the study (Bunting et al. 1980).

Much anecdotal evidence suggests that most ranchers in the southern Great Plains are not satisfied with the level of prickly pear control winter fires yield. This is especially true in areas where C_3 grasses, such as Texas wintergrass and annual bromegrasses (*Bromus* spp.), dominate. For these areas, it appears that summer fire may be necessary to gain effective control over pastures dominated by prickly pear cactus.

HERBACEOUS RESPONSES TO SUMMER FIRES

The literature is quite varied on assessment of individual grass species responses to summer-season fires. Part of the problem in interpretation arises from the differences in variables used to assess herbaceous responses. Many studies have taken an autecological approach and measured post-fire production of individual grass species over time (Reynolds and Bohning 1956, Cable 1967, Engle and Bultsma 1984, Whisenant et al. 1984). Those species that recover more slowly, usually in comparison to an unburned control, are generally viewed as fire intolerant. However, as mentioned earlier, this may be misleading because of the short-term nature of many fire studies. Other studies have measured ecosystem-level changes in species composition following fire (Steuter 1987, Biondini et al. 1989, Engle et al. 2000). Common variables used are species frequency of occurrence, richness, and/or percent basal cover. Those species that decreased relative to other species within the community matrix were viewed as less fire tolerant.

Sideoats Grama (C_4 Midgrass)

Ansley et al. (2006b) found in North Texas that sideoats grama end-of-growing-season total weight (live + dead) in the absence of grazing or clipping fully recovered from winter fires in 2 y, while it took 3 growing seasons to recover from summer fires (Figure 1). Fine fuel and peak fire temperature averaged 4,340 kg/ha and 718°C in winter and 4,205 kg/ha and 679°C in summer fire, respectively. Live yields recovered more rapidly than did total yields, and there was no difference in live yields between summer fire, winter fire, and the no-fire control by 2 y post-fire (data not shown). Annual spring clipping in addition to the fire treatments reduced total yields to a greater degree

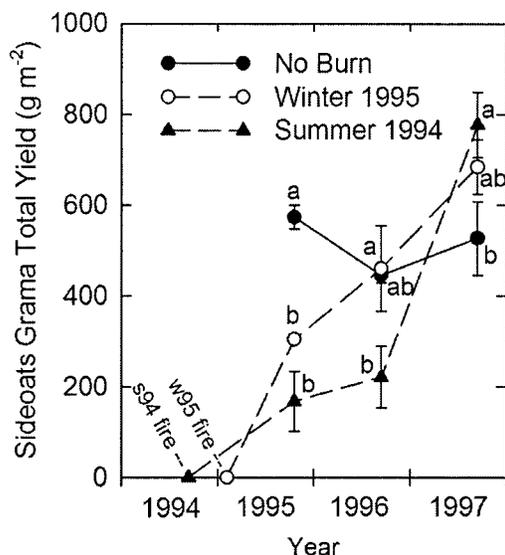


Fig. 1. Sideoats grama end-of-growing-season total standing crop following fires in summer 1994 (s94) or winter 1995 (w95) in North Texas. Means with similar letters within years are not significant ($P < 0.05$). Vertical bars are ± 1 SE. (Figure adapted from Ansley et al. 2006b.) Copyright © 2006 Society for Range Management. From *Rangeland Ecology and Management*, by R.J. Ansley, M.J. Castellano, and W.E. Pinchak. Reprinted by permission of Alliance Communications Group, a division of Allen Press, Inc.

in the no-fire and winter fire treatments than in the summer fire treatment when measured 1 y after clipping (Figure 2). At 5 y post-fire, a severe drought greatly reduced growth in the no-fire treatment, but high yields in both summer and winter fire treatments were maintained.

Wright (1974) found in West Texas that sideoats grama yields were reduced by 40–45% for the first 2 y after winter fires. These studies measured responses of the rhizomatous growth form of sideoats grama to fire, and Wright (1974:418) concluded that this form “never benefits from fire,” even if fires occur in the dormant season. Wright’s conclusions regarding sideoats grama responses to fire were based on 1–2 y of post-fire data. However, as stated previously, duration of post-fire measurements can affect interpretation of species tolerance to fire. Our study revealed that sideoats grama needed 3 y to recover fully and exceeded the no-fire control by 5 y post-fire (Ansley et al. 2006b). Taylor (2001) found in central Texas that sideoats grama frequency of occurrence increased to a greater degree 6 y after a summer fire than after either a winter fire or no fire. In New Mexico, summer fires increased sideoats grama cover by >100% (Brockway et al. 2002).

Several studies in the tallgrass prairie of the central Great Plains show a marked decline in C_4 midgrass and tallgrass production the first year after a summer fire but a recovery to pre-burn or unburned levels by the second or third year (Engle and Bultsma 1984; Ewing and Engle 1988; Engle et al. 1993, 1998; Engle and Bidwell 2001). In contrast, some C_4 bunchgrasses, such as threeawn (*Aristida* spp.), have elevated growing points and are killed by fire, especially summer

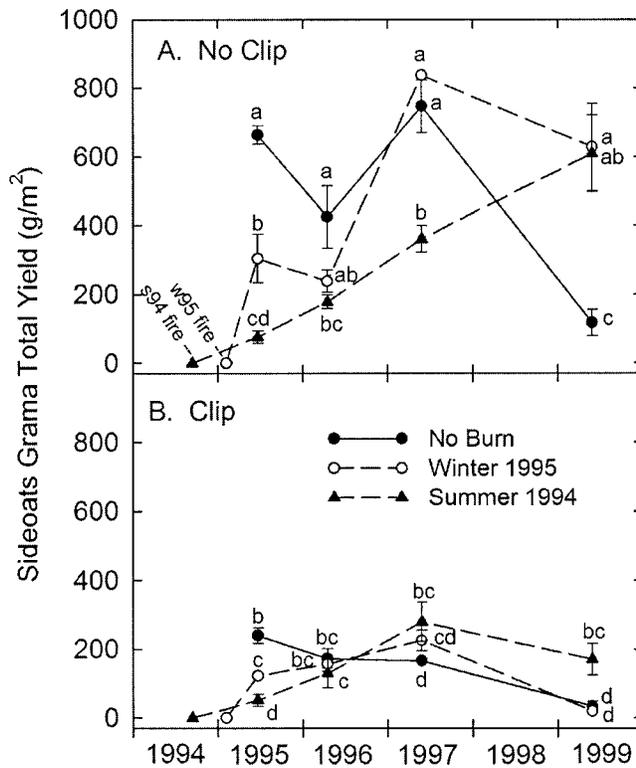


Fig. 2. Sideoats grama spring-season total standing crop following fires in summer 1994 (s94) or winter 1995 (w95) when unclipped (A) or clipped once each spring (B) in North Texas. Means with similar letters within years and across both panels are not significant ($P < 0.05$). (Figure adapted from Ansley et al. 2006b.) Copyright © 2006 Society for Range Management. From *Rangeland Ecology and Management*, by R.J. Ansley, M.J. Castellano, and W.E. Pinchak. Reprinted by permission of Alliance Communications Group, a division of Allen Press, Inc.

fires (Owensby and Launchbaugh 1977). In South Africa, Everson et al. (1985) and Trollope (1987) indicated that recovery of C_4 red grass (*Themeda triandra*) was significantly delayed by summer fire.

In summary, several studies have documented that C_4 midgrass and tallgrass species can survive summer fires and eventually fully recover from the disturbance. Recovery rates may be slower than what would occur following winter fires, although empirical evidence for this is not abundant.

Buffalograss (C_4 Shortgrass)

In a recent study in North Texas (Ansley and Castellano 2007b), buffalograss end-of-growing-season total weight in the absence of grazing or clipping recovered from both summer and winter fires the first growing season post-fire and exceeded the no-fire control by 3 y post-fire. Annual clipping once each spring in addition to either the summer or winter fire treatments did not have a negative effect on buffalograss yields when measured 1 y after clipping. The only time we have observed buffalograss to be negatively affected by summer fire was when it was exposed to 2 summer fires in consecutive years (with no clipping treatment). In this case, a community that was approx-

imately 60% buffalograss and 40% Texas wintergrass shifted to >90% Texas wintergrass (R.J. Ansley, unpublished data).

Other studies have found variable responses of buffalograss to summer fires. Buffalograss increased significantly following late-summer (September) fires on the Welder Wildlife Refuge in South Texas (Box et al. 1967). However, in New Mexico shortgrass prairie, summer fires reduced buffalograss cover from 5.7 to 0.5% (Brockway et al. 2002).

Other C_4 shortgrass species, such as blue grama (*Bouteloua gracilis*) and common curly-mesquite (*Hilaria belangeri*), appear tolerant of summer fires (Trlica and Schuster 1969, Mayeux and Hamilton 1988, Brockway et al. 2002). In contrast, in Southwest desert grasslands of New Mexico and Arizona, black grama (*Bouteloua eriopoda*) has been severely damaged by summer fire and may take up to 50 y to recover (Reynolds and Bohning 1956, Cable 1965, Drewa and Havstad 2001).

In summary, few studies have examined the effect of summer fire on buffalograss or other C_4 shortgrass species. However, the literature suggests that, with the exception of black grama, C_4 shortgrass species in general appear to be tolerant of summer fires and recovery rates are often more rapid than for C_4 midgrass species. However, it must be emphasized that very few studies have clearly documented this.

Texas Wintergrass (C_3 Midgrass)

In a recent study in North Texas, there were no negative long-term effects of winter or summer fires on Texas wintergrass total standing crop in the absence of grazing or clipping (Ansley and Castellano 2007b). The additional stress of annual spring clipping reduced total weights in the winter fire treatment, but not the summer fire or no-fire treatments when measured 1 y later. Thus, clipping (to simulate grazing) in spring exacerbated the negative effects of winter fire, but not summer fire.

Whisenant et al. (1984) found on one site in central Texas that Texas wintergrass standing crop was significantly reduced the first June following either a late-summer (September), winter (January), or late-winter (March) fire. By the second growing season post-fire, there was no difference in standing crop between fire and no-fire treatments. On a second site, Whisenant et al. (1984) found no differences in Texas wintergrass standing crop between the no-fire and the same 3 fire treatments either the first June or second growing season post-fire. In South Texas chaparral, Box and White (1969) found that Texas wintergrass herbage production was reduced to a greater degree by late-summer fires than by winter fires. Engle et al. (1998) found that two late-summer fires in tallgrass prairie increased Texas wintergrass production by 40% compared with no-fire controls on both a shallow and a deep soil site in Oklahoma.

In summary, with the exception of the study by Box and White (1969), Texas wintergrass appeared to be tolerant of summer fire with or without clipping to simulate grazing. The only time fire negatively affect-

ed Texas wintergrass was when spring clipping followed winter fire. Anecdotal observations often report an increase in dominance of Texas wintergrass following summer fires, and some studies support this observation. However, more research is needed to adequately document this response.

Herbaceous Composition Responses to Summer Fire

There is limited information on effects of summer fires on Great Plains herbaceous composition. Negative response to fire is hypothesized to increase if grass species are physiologically active at the time of burning (Daubenmire 1968, Howe 1994, Engle and Bidwell 2001). Thus, with respect to herbaceous species composition, in mixed stands of C_3 and C_4 herbaceous species, we might expect to see a shift toward C_4 species and away from C_3 species following winter or early spring fires, and the opposite following summer or early fall fires.

There is evidence that in northern and central Great Plains communities, spring fires will shift a mixed C_3 - C_4 grass community toward a greater C_4 presence (Anderson et al. 1970; Engle and Bultsma 1984; Steuter 1987; Howe 1994, 1995, 2000). There is less evidence that summer fires will shift mixed C_3 - C_4 communities toward a greater C_3 dominance. Steuter (1987) found in northern mixed prairie of South Dakota that summer fires shifted composition toward C_3 species, and Howe (1995) found in anthropogenically seeded C_3 - C_4 mixed-grass plots in Wisconsin that summer fires retarded C_4 grasses and favored C_3 species. However, in other studies in Wisconsin, Howe (1994, 2000) found that summer fires maintained a mix of C_3 and C_4 grasses, and thus increased diversity, but did not strongly favor C_3 grasses.

In the tallgrass prairie regions of Oklahoma, Engle et al. (2000) found that edaphic features and time since the last fire were the most important factors determining species composition on two Oklahoma prairie sites, but that summer fire did not necessarily cause long-term shifts in species composition. Ewing and Engle (1988) found in Oklahoma that a summer fire increased C_3 annual grass and decreased C_4 perennial grass production the first year post-fire, but long-term responses were not available. Coppedge and Shaw (1998) found in Oklahoma tallgrass prairie that summer fires increased C_3 sedges (*Carex* spp.), annual bromegrasses and forbs, and decreased C_4 tallgrasses and little bluestem (*Schizachyrium scoparium*) when compared with winter fires. These measurements were only made the first growing season post-fire, and long-term shifts in composition are not known. However, Engle and Bidwell (2001) concluded that summer fires do not cause long-term changes in species composition.

In South Texas, Owens et al. (2002) found no major shifts in species composition following summer fires. In North Texas, Texas wintergrass basal cover increased following summer fires but did not displace cover of C_4 grasses (R.J. Ansley, unpublished data). Texas wintergrass usually increased into areas that were either bare ground or undefined litter prior to the

fire. As indicated earlier, we found only one situation in North Texas, when summer fires were applied in two consecutive years, where Texas wintergrass actually replaced the dominant pre-fire grass, C_4 buffalo-grass (R.J. Ansley, unpublished data).

Numerous studies report increases in forb populations following fires. This appears to be true following repeated spring fires in northern mixed prairie (Biondini et al. 1989; Howe 1994, 1995), and in tall-grass prairie of eastern Kansas (Knapp et al. 1998), Oklahoma (Engle et al. 1993, 1998), and South Texas (Owens et al. 2002). Summer fires may drastically reduce grass production for several years (Engle and Bultsma 1984), but also can greatly increase subdominant species diversity, largely through increases in C_3 forbs (Biondini et al. 1989, Howe 1994, Drewa and Havstad 2001, Copeland et al. 2002). Summer fires appear to increase forbs to an even greater extent than do dormant-season fires (Biondini et al. 1989). For example, in North Texas, Tunnell and Ansley (1995) found much greater cover of common broomweed (*Amphiachyris dracunculoides*) the first growing season following summer (35%) than winter fires (3%). However, Box et al. (1967) found just the opposite result in South Texas: late-summer fires decreased forbs, while dormant-season fires increased forbs. The increase or decrease in forbs in response to summer fires may be viewed as a positive or negative response, depending on the desired land-use objective (Engle et al. 1993). Generally those with a wildlife management perspective would view an increase in forbs and species diversity as a positive response.

We are thus left with a mixed interpretation of summer fire effects on herbaceous composition but, in general, it appears that dormant-season fires are better able to shift C_3 - C_4 communities to C_4 dominance than are summer fires in shifting such communities toward C_3 dominance. Of those studies that document summer fires shifting species composition toward C_3 dominance, most have been located in the northern Great Plains (Steuter 1987; Howe 1994, 1995). This may simply mean that there are more studies of this nature in the northern Great Plains, or it may indicate that such a response occurs more readily in the northern Great Plains than in the southern Great Plains.

MANAGEMENT IMPLICATIONS

In all the research we have reviewed, most grass species appear to be tolerant of summer fire. An important exception is black grama in Southwest desert grasslands that appears to be very sensitive to fire, and some *Aristida* species. Prickly pear cactus is more susceptible to summer fires, and it is not difficult to imagine how the loss of summer fires in southern prairie ecosystems facilitated an increase in this species. Some woody species like honey mesquite may not be killed by fire, even after repeated summer fires of moderate to high intensity (flame heights 2–5 m) and burned with moderate herbaceous fine fuel loads (2,000–3,000 kg/ha). These fires successfully top-kill most mesquite,

but basal regrowth is stimulated. Fire must thus be applied repeatedly for sustained suppression of this species (Ansley and Jacoby 1998). Frequency of fire application depends on rate of woody regrowth but, typically, a fire every 7 to 10 y is considered adequate across most of the southern Great Plains (Scifres and Hamilton 1993). An important concept in fire application in mesquite-dominated systems is the necessity of managing livestock grazing to allow for an accumulation of an adequate herbaceous fine fuel amounts to facilitate the frequent use of fire (Scifres and Hamilton 1993, Teague et al. 1997).

Stewart et al. (2002:149) commented on how the loss of fire caused woody encroachment in Texas grasslands and how repeated prescribed fire was needed to restore such grasslands:

... grasslands were burned by Indians, and under the grazing by wild game such burning kept woody growth from becoming important. The early ranchers continued the Indian practice with the same results. With ever increasing herds of cattle and sheep, grasses were so reduced that the fires could no longer check the expansion of shrubs and trees. Without a thick stand of grass to carry the flames and produce sufficiently destructive heat, fires could no longer destroy brush. A very careful control of burning and grazing would be required to restore the grasslands to their pre-settlement condition and such control would have to be continued if fire were to serve the purpose it did for the Indians.

We would add that additional summer fires from lightning may have played a significant role in maintaining grasslands in the southern prairie and that, because of this, prescriptions for summer fires are necessary to aid in the grassland restoration process (Ansley and Taylor 2004). Further research is needed to identify safe and effective summer fire prescriptions and responses of other plant species as well as other components of the ecosystem, such as soil nutrients (Ansley et al. 2006a), before broad-scale recommendations for summer fire as a management practice can be made.

In addition, because no research to date has documented that fire in any season can kill a very high percentage of adult honey mesquite, it is our opinion that a few fire events alone cannot return heavily wooded stands of mesquite to grassland (Ansley and Castellano 2006). Thus, a repeated fire regime to maintain suppression of regrowth may be the best option unless an additional treatment, such as a herbicide or mechanical extraction, is employed. Because summer fires are inherently a greater risk to apply safely than are dormant-season fires (Ansley and Taylor 2004), one possible management scenario would be to initially apply a summer fire to top-kill a majority of the mesquite trees, assuming mesquite encroachment is not so high that it would prevent growth of an adequate herbaceous fine fuel, then follow up with a repeated fire regime that includes mostly dormant-season fires.

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EFFECTS OF DORMANT-SEASON FIRE AT THREE DIFFERENT FIRE FREQUENCIES IN SHORTGRASS STEPPE OF THE SOUTHERN GREAT PLAINS

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ABSTRACT

Prior to proceeding with large-scale fire reintroduction as a grassland management option, appropriate fire frequencies need to be determined. This research experimentally tested the effects of dormant-season fire on ground cover and on plant and soil nutrient cycling in shortgrass steppe at three different fire frequencies. The objective was to determine if fire return interval had detrimental effects on soil fertility and perennial grass cover relative to untreated grassland. This study is part of a long-term, 18-y study examining the effects of fire at return intervals of 3, 6, and 9 y. The study is located in the southern Great Plains of northeastern New Mexico (lat 36°31'20"N, long 103°3'30"W). The 160-ha site has mostly native vegetation, with the sod-forming *Buchloe dactyloides* and the bunchgrass *Bouteloua gracilis* being the dominant plant cover. The experimental design was completely randomized with four treatments and five replicate 2-ha plots per treatment. Treatments were fire return intervals of 3 y (3D) burned three times, 6 y (6D) burned twice, 9 y (9D) burned once, and unburned (U) plots. Treatments were applied in April 1997, 2000, and 2004. Vegetation and soil samples and ground-cover measurements were taken in October 2004. Response variables included percent litter, bare ground and live perennial grass cover, soil organic matter content and potentially mineralizable nitrogen, and *Bouteloua gracilis* nutrient content. Six months after fire treatments, there were no significant differences in perennial grass cover among treatments. However, litter cover was significantly reduced by increased burn frequencies ($P = 0.05$) from highest to lowest, as follows: U, 9D, 6D, and 3D. Conversely, bare ground increased with increased burn frequencies ($P = 0.05$) from lowest to highest, as follows: U, 9D, 6D, and 3D. Burn frequency had no significant effect on soil organic matter and nutrient availability. There were no significant differences in *Bouteloua gracilis* nutrient content, but there was a trend toward a higher percentage of nitrogen with more frequent burns. All fire frequencies produced minimal changes in grass cover and nutrient content and soil fertility. However, shorter fire return intervals may increase bare soil and erosion potential.

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FIRE ECOLOGY OF WOODY PLANT POPULATIONS IN UNGRAZED TALLGRASS PRAIRIE: EFFECTS OF SEASON OF FIRE ON DEMOGRAPHY, ABUNDANCE, AND REGENERATION

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ABSTRACT

Reductions in fire frequency and changing land use and management have enabled woody plant species to increase in grasslands worldwide. Nevertheless, fire is seldom eliminated from grasslands and the survival of woody shrubs depends on their fire-resistance traits and their ability to tolerate fire through regeneration from basal and aerial meristems. We assessed how season of fire affected growth, stem densities, and mortality rates of Jersey tea (*Ceanothus herbaceus*) and smooth sumac (*Rhus glabra*), two common shrubs of North American grasslands, in ungrazed sites. Additionally, we measured relative frequency and cover at various topographical positions on watersheds seasonally burned for 12 y, long-term unburned sites, and sites burned every 4 y. Although, spring, fall, and winter fires resulted in 100% stem mortality, fire stimulated increased basal resprouting, resulting in positive stem population growth rates. By contrast, populations in unburned and summer-burned sites showed lower mortality rates, both basal and aerial resprouting, but lower resprouting rates and lower population growth rates. Stem densities in spring-, fall-, and winter-burned sites increased significantly during the early growing season, but declined thereafter. Unburned areas did not show any change, while summer-burned populations showed continually decreasing stem densities. Therefore, our preliminary data suggest that summer burning may be a more effective management tool in reducing shrub densities and population growth rates in grasslands than spring burning, the most common management practice in the Flint Hills region. However, summer watersheds are only burned every 2 y because of insufficient fuel loads and these data may not show a full oscillation of stem-density dynamics. Genet (clone) populations were characterized by stable densities and low flux, whereas intraclonal ramet (stem) densities showed high turnover rates with fire, suggesting that frequent fire strongly influences the demography but not the genetic structure of clonal shrub populations.

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RESTORING BARRENS SHRUBLANDS: DECREASING FIRE HAZARD AND IMPROVING RARE PLANT HABITAT

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ABSTRACT

The coastal sandplain of southeastern New England is characterized by a complex of xeric, fire-prone shrubland and open woodland communities. Also known as barrens, these landscapes supported frequent fires until recently when suppression efforts became effective. Currently, shrubland components are increasingly being lost through succession to closed woodlands. Within our study area—the Manuel F. Correllus State Forest (MFCSF) on the island of Martha's Vineyard—heathlands and scrub oak (bear oak [*Quercus ilicifolia*] and dwarf chinkapin oak [*Q. prinoides*]) shrublands are declining while oak (*Quercus* spp.) woodlands (and less frequently pitch pine [*Pinus rigida*] forests) increase in area.

The MFCSF, with 11 rare plant species, has one of the highest concentrations of rare plants in the coastal region. Our goal was to determine if management (including thinning, brushcutting, grazing, pile burning, and broadcast prescribed burning) can be used both to reduce fire hazard and create rare plant habitat. Species composition was largely unchanged by treatments, as many species resprouted. However, following fuel treatments, new occurrences of two rare species and several grassland-associated species were found where bare soil was created. Grassland associates colonized plots with lower duff depths and greater lichen cover than plots lacking these species. Grazing increased establishment of nonnative species, although most did not persist past the first growing season.

Treatments reduced fine (1- and 10-h time lag) fuel loads >50% from averages of 32 t/ha and 23 t/ha in scrub oak and pitch pine, respectively. In scrub oak, treatments reduced shrub fuel depths >75% (from 1.3 to 0.3 m). In untreated plots (under moderate midflame wind speeds of 5.8 km/h and humidities of 65%), rates of spread of research burns were >4.6 m/min and flame lengths were >2 m. Under similar burning conditions in treated plots, flame lengths were <0.6 m and rates of spread <2.1 m/min. Initial results suggest that treatments created rare plant habitat and reduced fire hazard without adversely impacting native shrub communities.

keywords: Atlantic Coastal Plain, barrens, fire hazard, fire intensity, fuel loads, fuel management, Massachusetts, rare plants, shrublands.

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INTRODUCTION

In the northeastern United States, open, early seral communities are rare and declining in size and number (Noss et al. 1995, Motzkin and Foster 2002). Most occur in sandplains with level topography and coarse-textured soils. They support a variety of plant communities, including grasslands, heathlands, shrublands, oak (*Quercus* spp.) savannas, and pitch pine (*Pinus rigida*)–scrub oak (bear oak [*Quercus ilicifolia*] and dwarf chinkapin oak [*Q. prinoides*]) barrens (Goldstein 1997). Sandplain soils, for the most part derived from outwash left by retreating ice of the Wisconsin glacial stage, are well drained and nutrient poor. The unique physical and phytosociological characteristics of sandplains combine to create ecosystems that are globally rare. Often referred to as *barrens*, they provide habitat for numerous rare plant and animal species, making them the focus of conservation efforts (Noss et al. 1995, Barbour et al. 1998, Massachusetts Natural Heritage and Endangered Species Program 2001, Wagner et al. 2003).

Barrens vegetation produces abundant, flammable

fuel that decomposes slowly, creating the potential for intense wildfires. Sandplains are preferred sites for housing and light industry, which increases the need for fuels management to reduce wildfire threat to human resources. Fuel breaks (areas with reduced fuels) have been created and maintained at barrens sites in Massachusetts, using a variety of techniques including thinning, mowing, grazing, plowing, and, to a limited extent, prescribed burning. In some areas, such as the Manuel F. Correllus State Forest on the island of Martha's Vineyard, these fuel breaks are dominated by grassland vegetation and currently support populations of state-listed rare plants. However some consider these fuel breaks to be unnatural because of the plowing and mowing required for their creation and maintenance. Alternative management practices that do not create grasslands might be used to reduce fuels and maintain open sandplain habitats more typical of natural conditions (Foster and Motzkin 1999).

In 2002, we began a 3-y study of alternative methods for fuel break maintenance and expansion, including overstory thinning, understory mowing and grazing, and prescribed burning (Patterson et al. 2005). To determine the effects of different management options on rare plant habitat and to demonstrate the extent to which alternative management techniques affect fuel beds and fire behavior in the three major sandplain

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Fig. 1. Location of Martha's Vineyard on Massachusetts' south coast.

vegetation types—pitch pine, oak woodland, and scrub oak—we established 27 experimental treatment plots in which we compared vegetation composition, fuel bed characteristics, and prescribed fire behavior in treated and untreated plots.

STUDY AREA

One of the largest undeveloped sandplains in Massachusetts was in the Manuel F. Correllus State Forest, located at the center of the 256-km² island of Martha's Vineyard—9.6 km south of Cape Cod (41°23'N, 70°39'W; Figure 1). Nearly 1,670 ha of barrens vegetation remained in an area recognized as critical habitat for several plant species that were rare in New England (Table 1). One of these—sandplain gerardia (*Agalinis acuta*)—is federally endangered. More than 20 rare moths (Lepidoptera) were also recorded for the forest. Some of these rare moth species represented the only New England populations ever recorded and may be disjunct populations of prairie species that were more common in the Midwest (Goldstein 1997, Mehrhoff 1997).

The state forest was underlain by highly permeable soils derived from outwash and morainal deposits (Fletcher and Roffinoli 1986). Topography of the area was relatively flat, with the elevation not varying more than 12.2 m. Slopes were generally <5%. The climate of the island was humid continental, with average monthly precipitation ranging from a high of 12 cm in

Table 1. Rare plants evaluated in the Manuel F. Correllus State Forest, Martha's Vineyard, Massachusetts, 2002–2004.

Species	State status
Purple needle-grass (<i>Aristida purpurascens</i>)	Threatened
Sandplain flax (<i>Linum intercursum</i>)	Special concern
Lion's foot (<i>Nabalus serpentarius</i>)	Endangered
Papillose nut-sedge (<i>Scleria pauciflora</i> var. <i>caroliniana</i>)	Endangered
Sandplain blue-eyed grass (<i>Sisyrinchium fuscatum</i>)	Special concern

November to a low of 7.4 cm in July (Stormfax 2005). Humidity was typically high and temperatures were moderated by the surrounding sea. Days with low humidities occurred in the spring, often with high winds (Mouw 2002).

The earliest quantitative description of the forest's vegetation was from interpretation of aerial photos taken in the 1930s, more than 250 y after European colonization (Mouw 2002). In the early 20th century, the area was dominated by immature tree and shrub oaks—the product of many decades of cutting and wildfire (Foster and Motzkin 1999). Only in the past 5 decades have efforts to suppress fires been effective. The modern fire regime is characterized by small (generally <4 ha [10 acres]; infrequently >45 ha [100 acres]) fires with return intervals of a decade or longer (Mouw 2002). The most abundant vegetation types on the forest were scrub oak (29% of the forest), oak woodlands (26%), mixed-oak woodland–scrub oak (17%), and plantations (16%) (Mouw 2002). Pitch pine stands composed <5% of the vegetation. The remaining approximately 7% of the forest was grasslands and heathlands on a grid of firelanes, which dissected the forest into blocks generally 0.8 km (0.5 mi) on a side (Clarke and Patterson 2007). These firelanes were established in the early 20th century in an initial attempt to break up fuels and prevent the spread of fires to the private land surrounding the forest (Foster and Motzkin 1999, Mouw 2002). Originally referred to as *fuel breaks*, we describe them as *firelanes* to acknowledge that they are not wide enough (generally <30–60 m) to stop a running headfire. They do, however, improve access for suppression crews.

METHODS

Southwest Experimental Fuel Break

We established a fuel break in the southwest corner of the forest, and within this fuel break established plots on which different fuels management techniques were applied (Figure 2). Our goal was to evaluate the effects of different management practices on fuel bed characteristics, fire behavior, and vegetation. Simulations showed that fires ignited in this area have the greatest potential to burn large areas under worst-case conditions (Mouw 2002). Fires burning with a southwest wind can run for 1.6 km or more through largely untreated fuels before reaching the northern boundary of the forest. The 152-m (500-foot) wide and nearly

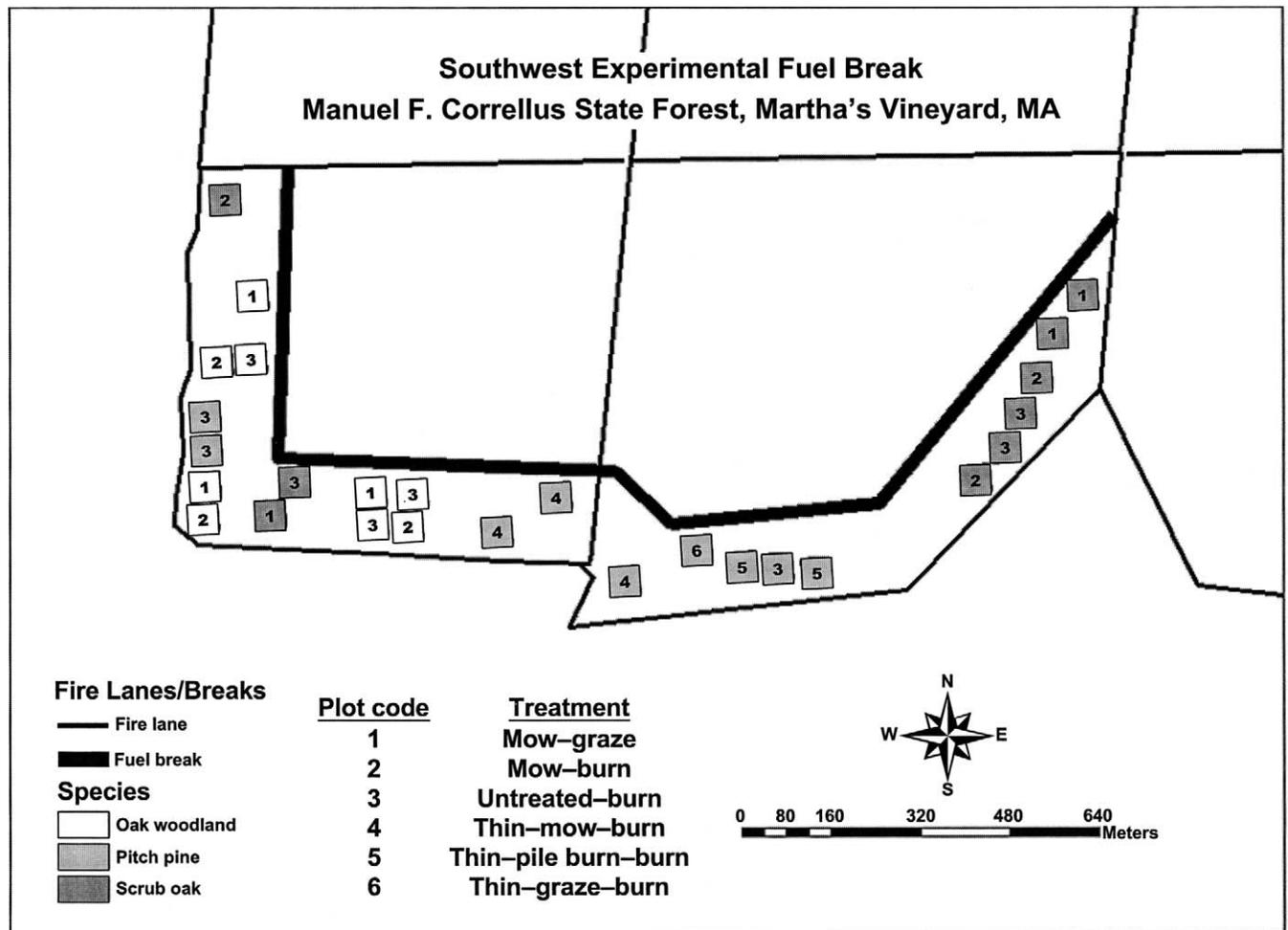


Fig. 2. Layout of 27 40 × 40-m experimental fuel treatment plots in three vegetation types (oak woodland, pitch pine, and scrub oak) in the southwest corner of the Manuel F. Correllus State Forest on Martha's Vineyard, Massachusetts, 2002–2004. Vegetation types are as described by Mouw (2002). See Table 2 for individual plot treatments. Map prepared by Dustin Parker.

3.2-km (2-mi)-long fuel break includes stands of oak woodland, scrub oak, and pitch pine.

Treatments

Within the fuel break, treatment plots to evaluate the effectiveness of different fuel management techniques were created in three fuel types (pitch pine, oak woodland, and scrub oak) with three treatment combinations (overstory thinning [pitch pine] and understorey mowing and/or sheep grazing, plus untreated controls) replicated three times per fuel type, for a total of 27 plots (see Table 2 for details). Fuel types were distinguished by canopy cover (pitch pine: >25% pitch pine, <25% tree oak; oak woodland: oaks dominant with <5% other species and ≥25% tree cover; scrub oak: <25% tree cover). Treatments were located on replicated, 45 × 45-m (0.2-ha [0.5-acre]) plots.

Following treatments, all 27 plots were burned between 29 April and 7 May 2004 to document differences in fire behavior between treated and untreated conditions. To complete the experimental fuel break and facilitate burning the treatment plots, the area

around the plots and within the fuel break was also burned during February–March 2004.

Vegetation Sampling

We sampled all treatment plots with relevés between early July and mid-August 2002, prior to application of treatments. In 2003, relevé sampling was repeated in plots where treatments had occurred. Mow-graze plots were sampled twice: just before and just after grazing. At the end of the final (2004) field season, all treatment plots were resampled. Surveys for rare plant species were conducted on individual treatment plots and throughout the entire experimental fuel break at the beginning (2002) and end (2004) of the study.

To illustrate the extent and direction of vegetation change following treatments, nonmetric multidimensional scaling ordination was conducted using PC-ORD 4.0 (McCune 1993) with Sorensen's distance measure. Importance values (Clark and Patterson 1985) were calculated for each species from abundance values across strata. Species occurring in fewer

Table 2. Treatment history for individual plots (9 per cover type; 27 total), showing month(s) of treatment in parentheses, Manuel F. Correllus State Forest, Martha's Vineyard, Massachusetts, 2002–2004.

Cover type	Treatment	Plot no.	Treatment		
			2002	2003	2004
Pitch pine	Control	1,5,9	None	None	Burn (Apr, May)
	Thin–mow	2,3,7	Thin (Jul)–mow (Jul)	None	Burn (May)
	Thin–pile burn ^a	4,6	None	Thin (Aug)–pile burn (Jan, Feb)	Burn (May)
	Thin–graze–pile burn ^a	8	Thin(Jul)–graze (Aug)	Pile burn (Jan, Feb)	Burn (May)
Oak woodland	Control	2,7,8	None	None	Burn (Apr)
	Mow–graze	1,5,9	Mow (Jul)	Graze (Jun, Jul, Aug)	Burn (Apr)
	Mow	3,4,6	Mow (Jul)	None	Burn (Apr)
Scrub oak	Control	4,5,8	None	None	Burn (May)
	Mow–graze	1,2,9	Mow (Jul)–graze (Aug, Sep)	Graze (Sep)	Burn (May)
	Mow	3,6,7	Mow (Jul)	None	Burn (Apr, May)

^a Grazing on pitch pine plots was abandoned after one unsuccessful season on pitch pine plot no. 8.

than two plots were removed from the data set prior to analysis. Two-sample tests of variances and *t*-tests were run in SAS 9.1 (SAS Institute 2004) to evaluate the effects of treatments on environmental variables.

Rare Plant Habitat Sampling

Rare plants were surveyed throughout the forest. Relevé (Mueller-Dombois and Ellenberg 2002) sampling and sampling for environmental variables (point-intercept sampling for ground cover and vegetation structure) were conducted on 100-m² plots where rare plant species were found (Clarke and Patterson 2007). Sampling was also conducted on 100-m² plots at randomly located points in each of the five native cover types identified by Mouw (2002): pitch pine, oak woodland, oak woodland–scrub oak, scrub oak, and grassland. A total of 64 random plots off of firelanes and 54 within firelanes were sampled in 2003 and 2004. This effort allowed us to determine which species and habitat variables were most likely associated with rare plants. Systematic searches were also made within the entire experimental fuel break and within each of the 27 0.2-ha treatment plots during late summer 2004, after all treatments had been applied.

Fuels and Fire Behavior

Fuels on all treatment plots were sampled at the outset during summer 2002 and after mechanical treatments during summer 2003. All plots, including controls, were sampled prior to burning in spring 2004 and again in August 2004. Sampling within plots included downed woody fuel lines (Brown 1974), point-intercept lines for vegetation and ground cover, canopy cover (spherical densiometer), tallies of scrub oak stems (by 0.25-cm-basal-diameter classes) on 1-m² subplots (omitted in spring of 2004). Litter, downed wood, and live and dead standing fuels were harvested from 1,600-cm² subplots. One-hour time-lag fuels (those <0.6 cm diameter) were manually separated from 10-h fuels (those 0.6–2.5 cm diameter).

Rates of spread and flame lengths were estimated by comparisons with 1.3-m-tall reference poles and video-recorded several times on each treatment plot during all prescribed burns. BehavePlus (Andrews et al. 2005) was used to develop custom fuel models for

each vegetation–treatment combination using fuel measurements from individual plots, averaged across replicates of treatment types within vegetation types (see www.umass.edu/nrc/nebarrensfuels/ for data summarization techniques). Time-of-burn wind speed and fuel moisture data were used with custom fuel models (CFMs) to predict fire behavior in each plot using NEXUS (Scott 1999). Pearson's product–moment correlations (Sokal and Rohlf 1995) were used to compare predicted and observed flame lengths and rates of spread on representative plots for each fuel type–treatment combination.

RESULTS

Treatment Response

For the 27 treatment plots within the experimental fuel break, ordination (Figure 3) readily distinguished the three main vegetation–fuel types: oak woodland, scrub oak, and pitch pine. Although the treatments altered the structure of vegetation on the plots, changes did not warrant their reclassification to a different vegetation type. A number of plots shifted with mechanical treatments, especially those in pitch pine, toward increased importance of grassland associates (which have high axis 1 and intermediate axis 2 scores on Figure 3). Treatments did not substantially alter overall species composition; most species on these plots resprouted, and by 2004 many had importance values that equaled or exceeded pretreatment values. However, a number of grassland associates colonized experimental plots, especially in scrub oak and pitch pine (Table 3)—in the latter especially where bare soil was exposed by machinery and/or pile-burning. Those plots colonized by more grassland associates shifted farther to the right along axis 1 in the ordination (e.g., PP8; Figure 3). Several nonnative grass species (e.g., common velvetgrass [*Holcus lanatus*], timothy [*Phleum pratense*]; Patterson et al. 2005) were introduced in grazed plots (particularly in the first plot grazed each season), but few persisted past the first growing season.

Treatments minimally impacted duff depths but generally reduced average litter depth (significantly in scrub oak mow–graze and all pitch pine plots). How-

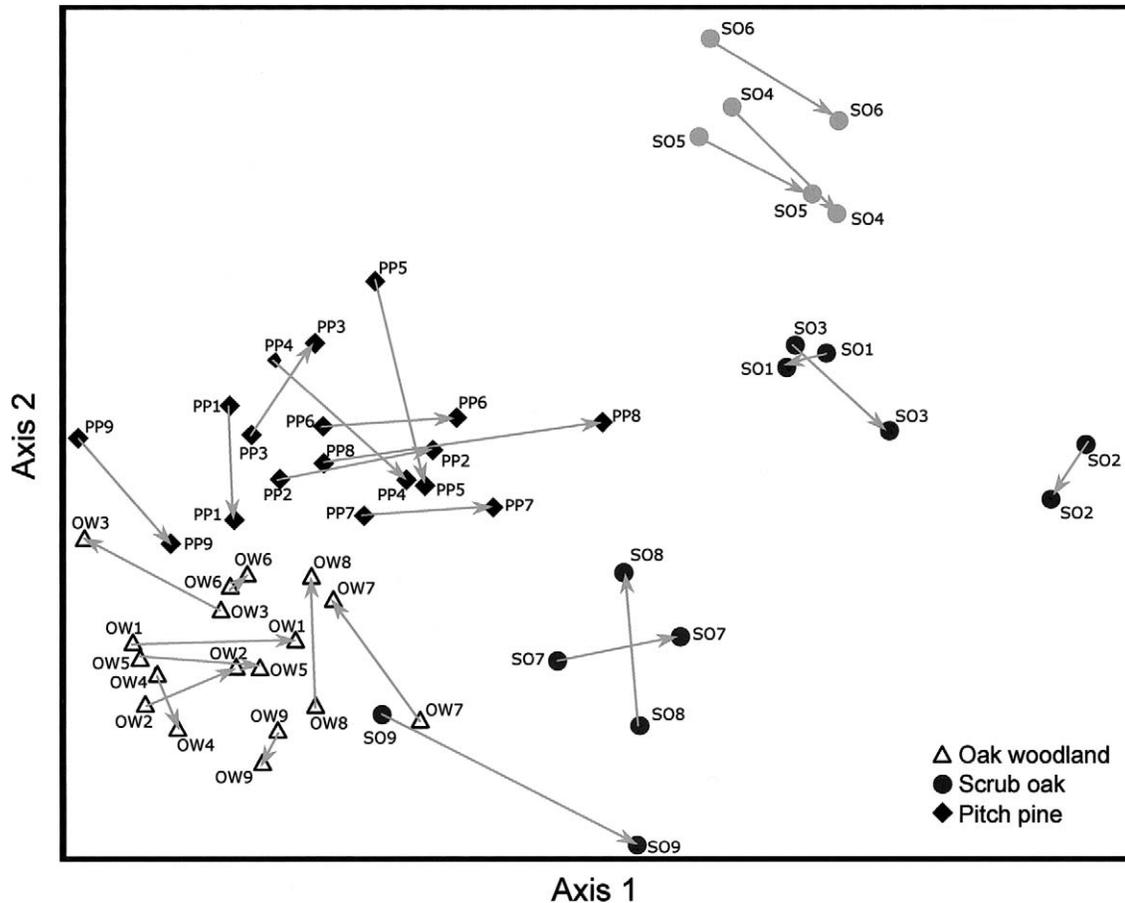


Fig. 3. Ordination (nonmetric multidimensional scaling) of 27 experimental fuel treatment plots in space defined by species abundances (importance values) in 2002 and 2004 (arrows point toward the latter), Manuel F. Correllus State Forest on Martha's Vineyard, Massachusetts, 2002–2004. Grassland-associated species have high axis 1 scores and intermediate axis 2 scores. Treatments generally shifted scrub oak (SO) and pitch pine (PP) plots toward increasing importance of grassland associates, with no consistent trends identified in the movement of oak woodland (OW) plots.

ever, treatments often resulted in significant increases in litter load due to the addition of shredded shrub stems to the litter layer.

Rare Plants

In searches throughout the forest, rare plants were only found in disturbed areas, either in firelanes or along paths where frequent disturbance occurs. However, grassland associates, which occurred often (>80% frequency of occurrence) in firelane plots and rarely in woodland and shrubland plots (<10% frequency of occurrence), were found, where they did occur in woodlands–shrUBLANDS, in association with significantly lower duff depth and higher lichen cover than is typically found on woodland and shrubland plots (Clarke and Patterson 2007). These grass and herb species occurred occasionally in plots characterized as oak woodland–scrub oak (3 of 29 plots), pitch pine (2 of 15 plots), and oak woodland (1 of 12 plots), and frequently in scrub oak (4 of 7 plots).

During pre-treatment sampling within the experimental fuel break, only one stem of one rare plant species (sandplain blue-eyed grass [*Sisyrinchium fuscatum*]) was found along a bridle path. Following treatments, papillose nut-sedge (*Scleria pauciflora* var.

caroliniana) established at two sites in areas where bare soil was exposed by machinery, and lion's foot (*Nabalus serpentarius*) was found in a scrub oak stand that had been mowed and burned.

Treatments affected environmental variables in ways that might be interpreted as being favorable to rare plants. Vegetative (canopy) cover >0.5 m in height in scrub oak plots was equivalent to that found on rare plant plots prior to, and was further reduced by, treatments (Table 4). Canopy cover was significantly higher in pitch pine and oak woodland plots (Tables 5, 6) than in rare plant plots both before and after treatments, despite a significant reduction in cover following thinning in pitch pine plots. Litter depths were reduced by treatments in all three vegetation types (Tables 4–6), although duff depths remained significantly higher than in rare plant plots (Figure 4).

Fuels and Fire Behavior

In the first and second growing seasons after treatments, shrub fuel loads were generally reduced to <50% of pretreatment levels, and shrub heights to <25% (Figure 5). Shrub heights in oak woodland plots were more heterogeneous because mowing was inten-

Table 3. Number of research plots by treatment and vegetation type in which grassland-associated species established following treatments, Manuel F. Correllus State Forest, Martha's Vineyard, Massachusetts, 2002–2004. Treatments are control (C), thin (T), mow (M), graze (G), and pile burn (P). In addition, all plots were broadcast burned in April–May 2004. Number of burn piles indicates the number of burn piles the species was found to have established within (out of 24). No grassland associates established in oak woodland–mow, pitch pine–control, or in scrub oak–control.

Species	No. of burn piles	Oak woodland		Pitch pine		Scrub oak	
		C	MG	TP	TM	M	MG
Bentgrass (<i>Agrostis</i> sp.)				1		1	3
Pearly everlasting (<i>Anaphalis margaritacea</i>)			1				1
Late purple aster (<i>Aster patens</i>)							1
Showy aster (<i>Aster spectabilis</i>)						1	
Poverty grass (<i>Danthonia spicata</i>)	1					1	
Common hairgrass (<i>Deschampsia flexuosa</i>)						3	1
Poverty panic grass (<i>Dichanthelium depauperatum</i>)	1	1	2	3	3	2	3
Cypress panic grass (<i>Dichanthelium dichotomum</i>)	4					2	2
Slender or grass-leaved goldenrod (<i>Euthamia</i> sp.)				1		1	1
Fescue (<i>Festuca</i> sp.)				1			
Frostweed (<i>Helianthemum</i> sp.)	4		1	2	2		
Pinweed (<i>Hypericum gentianoides</i>)				1			1
Pinweed (<i>Lechea minor</i>)	2			3	2	1	1
Bush-clover (<i>Lespedeza</i> sp.)			1				
Whorled loosestrife (<i>Lysimachia quadrifolia</i>)		2	2	2	2	1	2
Nuttall's milkwort (<i>Polygala nuttallii</i>)						1	
Cinquefoil (<i>Potentilla</i> sp.)						1	1
Little bluestem (<i>Schizachyrium scoparium</i>)						2	1
Sweet goldenrod (<i>Solidago odora</i>)		1				2	
Downy goldenrod (<i>Solidago puberula</i>)						2	
Rough-stemmed goldenrod (<i>Solidago rugosa</i>)				1		1	1
Birdfoot violet (<i>Viola pedata</i>)							1

tionally done in patches (Figure 5). Prior to treatments, scrub oak had higher 1- and 10-h fuel loads (32 t/ha) than pitch pine (24 t/ha) or oak woodland (23 t/ha). Reductions in fuel loads with treatments were generally greatest in scrub oak where shrub loads were reduced to approximately 30% of pre-treatment values in mowed plots and to 10% in mow–graze plots. Litter loads generally increased with treatment, frequently by >10%, whereas litter depths generally decreased (with significant decreases in scrub oak and pitch pine plots [Tables 4, 5]), reflecting the compaction of litter by treatments.

Without treatment, observed fire behavior was extreme (Figures 6, 7). With moderate weather conditions (average wind speeds of 5.8 km/h [3.6 mi/h] and average relative humidity of 65%), flame lengths averaged 2.5 m (8.2 ft) and rates of spread >5.8 m/min (19 ft/min). In all three cover types following treatments, flame lengths averaged <0.5 m (1.6 ft) and rates of spread <1 m/min (3.3 ft/min). Differences be-

tween treatments were small. Many treated plots did not burn completely (Table 7) because treatments reduced the horizontal continuity of fuels.

Custom fuel models run in NEXUS with weather data collected at the time of prescribed burns predicted fire behavior similar to that observed, with nearly 1:1 correlations between observed and predicted values (flame lengths, $r = 0.90$; rates of spread, $r = 0.95$ [Figures 6, 7]). Observed flame lengths were generally somewhat lower than predicted except in scrub oak (all treatments) and pitch pine (thin–mow), where they were somewhat higher. Rates of spread were generally somewhat underpredicted, except in treated oak woodlands.

DISCUSSION

Impacts of Management on Rare Plant Habitat

The colonization of grassland associates (and in a few cases rare species) in experimental plots following

Table 4. Environmental variables sampled in scrub oak treatment plots and rare plant plots containing any of five rare plant species with averages and confidence interval half-widths ($P = 0.05$), Manuel F. Correllus State Forest, Martha's Vineyard, Massachusetts, 2002–2004. Pre-treatment values are for plots sampled in 2002 and treatment values are for plots sampled in the spring of 2004, after treatments but before prescribed burns. Plot means for each variable that are not significantly different are followed by the same letter (t -test).

Variable	Treatment plots			Rare plant plots
	Pre-treatment	Post-mow	Post-mow/graze	
Litter depth (cm)	7 ± 2a	2 ± 0b	2 ± 3b	1 ± 0b
Bare soil cover (%)	1 ± 1a	0 ± 1a	1 ± 2a	10 ± 5b
Grass cover (%)	1 ± 2a	3 ± 11a	2 ± 3a	58 ± 7b
Herb cover (%)	0 ± 1a	0 ± 1a	0 ± 0a	22 ± 4b
Woody cover (%)	98 ± 1a	73 ± 12b	68 ± 13b	39 ± 6c
Vegetation cover >0.5 m (%)	15 ± 15b	10 ± 30b	5 ± 20b	20 ± 5b

Table 5. Environmental variables sampled in pitch pine treatment plots and rare plant plots containing any of five rare plant species with averages and confidence interval half-widths ($P = 0.05$), Manuel F. Correllus State Forest, Martha's Vineyard, Massachusetts, 2002–2004. Pre-treatment values are for plots sampled in 2002 and treatment values are for plots sampled in the spring of 2004, after treatments but before prescribed burns. Plot means for each variable that are not significantly different are followed by the same letter (t -test).

Variable	Treatment plots			Rare plant plots
	Pre-treatment	Post-thin/mow	Post-thin/pile burn	
Litter depth (cm)	11 ± 3a	3 ± 3b	4 ± 2b	1 ± 0b
Bare soil cover (%)	0 ± 1a	0 ± 0a	1 ± 3a	10 ± 5b
Grass cover (%)	0 ± 0a	0 ± 0a	1 ± 6a	58 ± 7b
Herb cover (%)	0 ± 0a	0 ± 0a	0 ± 1a	22 ± 4b
Woody cover (%)	99 ± 2a	58 ± 11b	57 ± 34bc	39 ± 6c
Vegetation cover >0.5 m (%)	65 ± 10a	50 ± 20a	50 ± 20a	20 ± 5b

treatments suggests that fuel reduction treatments improved habitat for rare plants. Establishment of some species, including the rare papillose nut-sedge, was apparently dependant on the exposure of mineral soil. Deep organic layers may prevent germination of graminoid and herbaceous seed, and may make grassland associates good indicators of favorable habitat for rare plants (Clarke and Patterson 2007). Average duff depths did not decrease significantly following treatments, but patches of exposed mineral soil were found in some mechanically treated plots, and grassland associates colonized these patches.

Scrub oak plots had thinner duff layers than other vegetation types prior to treatments, and both mowing and grazing increased establishment by grassland associates in scrub oak despite increased litter loads due to mowing. Duff and litter loads remained high in oak woodlands and pitch pine stands, and grassland associates did not readily colonize following treatments. However, grasses and forbs colonized sites where mineral soil was exposed in small patches by slash pile burning and disturbance by tree-felling equipment in pitch pine stands.

Thinning and prescribed burning may have benefited existing rare plant populations. Lion's foot, which is easily overlooked when not flowering, was probably present prior to treatments but was not discovered until it flowered following treatments. The extent to which reduction in canopy cover due to thinning improved habitat conditions for this and other rare species is not easily separated from the effects of soil disturbance because they colonized thinned plots where bare soil was exposed by machinery. A few unthinned oak

woodland plots were colonized by grassland associates following mowing (Table 3). Thus, a lack of soil disturbance may be more important than shading in preventing establishment of grassland associates in woodlands. Several studies suggest that increasing shrub cover in open sandplains decreases grass and herb richness (Dunwiddie and Caljouw 1990, Harper 1995), although no studies have differentiated the influence of shading from soil disturbance.

Establishment by grassland associates and rare species in the experimental fuel break may be the result of recruitment either from the seed bank or from propagules dispersed from outside the break (i.e., from nearby seed sources in firelanes that border the fuel break), possibly dispersal by machinery. Elsewhere on Martha's Vineyard, Lezberg et al. (2006) found few grassland species establishing following clearcutting and mowing in an oak woodland community. Matlack and Good (1990) found little seed banked in the soil of a New Jersey coastal plain forest, but nut-sedge does appear to bank seed (Oosting and Humphreys 1940, Clinton and Vose 2000, Zaremba 2003).

We found several grassland associates emerging from the ash under burned slash piles. The extended residence time that occurs when burning piles may stimulate soil-banked seeds of these species to germinate. Pile burning has been documented as having both positive and negative effects on banked seeds (e.g., Clark and Wilson 1994) and can encourage the establishment of nonnative species (Haskins and Gehring 2004, Korb et al. 2004), although we found no such evidence.

Table 6. Environmental variables sampled in oak woodland treatment plots and rare plant plots containing any of five rare plant species with averages and confidence interval half-widths ($P = 0.05$), Manuel F. Correllus State Forest, Martha's Vineyard, Massachusetts, 2002–2004. Pre-treatment values are for plots sampled in 2002 and treatment values are for plots sampled in the spring of 2004, after treatments but before prescribed burns. Plot means for each variable that are not significantly different are followed by the same letter (t -test).

Variable	Treatment plots			Rare plant plots
	Pre-treatment	Post-mow	Post-mow-graze	
Litter depth (cm)	6 ± 2a	3 ± 2ab	2 ± 1b	1 ± 0b
Bare soil cover (%)	0 ± 0a	0 ± 0a	0 ± 0a	10 ± 5b
Grass cover (%)	0 ± 0a	0 ± 1a	0 ± 0a	58 ± 7b
Herb cover (%)	0 ± 0a	0 ± 0a	0 ± 0a	22 ± 4b
Woody cover (%)	99 ± 1a	88 ± 7b	83 ± 25ab	39 ± 6c
Vegetation cover >0.5 m (%)	52 ± 8a	66 ± 18a	54 ± 8a	20 ± 5b

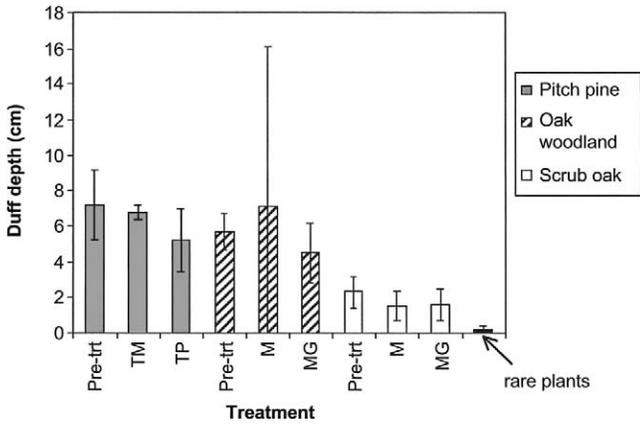


Fig. 4. Average duff depths in fuel treatment plots, by vegetation type, in an experimental fuel break prior to and following treatments (including confidence intervals [$P = 0.05$]), Manuel F. Correllus State Forest on Martha's Vineyard, Massachusetts, 2002–2004. Pre-treatment (Pre-trl) values are the average of six plots sampled in 2002. Treatment values are the averages of three plots sampled in the summer of 2004. Also shown is the average value in firelane plots supporting any of five rare plant species. Treatment codes: C, control; G, graze; M, mow; P, pile burn; T, thin.

Fuels and Fire Behavior

In all three vegetation types, average headfires burning through untreated fuels with moderate weather conditions cannot be attacked directly with hand tools (flame lengths >1.2 m [4 ft]). In scrub oak, where fine fuel loadings and depths are high, we observed typical flame lengths of 3.5 m (11.4 ft) and rates of spread of 4.8 m/min (15.7 ft/min). Woodall (1998) reported extreme behavior in an early growing-season burn (75% humidity with wind speeds of 3.2–6.4 km/h [2–4 mi/h]) in Martha's Vineyard scrub oak, with rates of

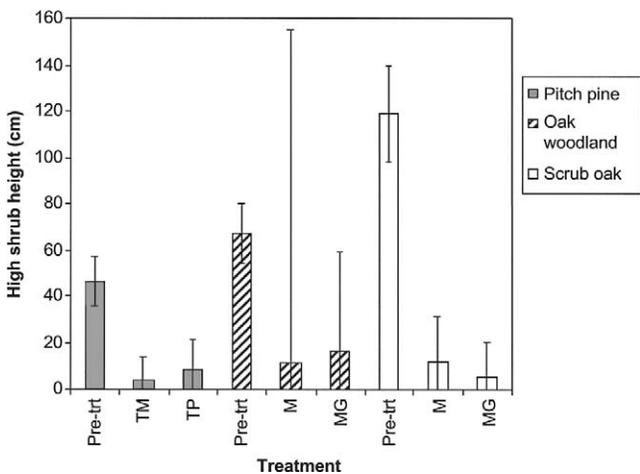


Fig. 5. Average shrub heights in fuel treatment plots, by vegetation type, in an experimental fuel break prior to and following treatments (including confidence intervals [$P = 0.05$]), Manuel F. Correllus State Forest on Martha's Vineyard, Massachusetts, 2002–2004. Pre-treatment (Pre-trl) values are the average of six plots sampled in 2002. Treatment values are the averages of three plots sampled in the spring of 2004. Treatment codes: C, control; G, graze; M, mow; P, pile burn; T, thin.

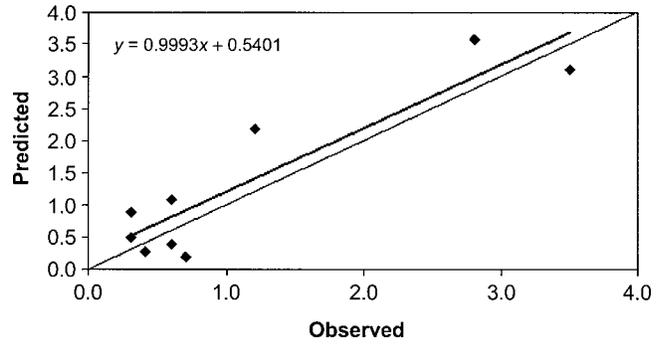


Fig. 6. Predicted versus observed flame lengths (m) for prescribed burns in control and fuel treatment plots, Manuel F. Correllus State Forest on Martha's Vineyard, Massachusetts, 2002–2004. The 1:1 diagonal line represents a perfect fit. Observed values for treatment plots are all ≤ 0.7 m.

spread of 18.2 m/min (60 ft/min) and flame lengths of 12.2 m (40 ft).

Custom fuel models are clearly useful for barrens in the northeastern United States, as standard fuel models either under- or overpredict observed fire behavior, whereas our custom fuel models predicted observed behavior well. Predicted flame lengths were within 1.1 m (3.6 ft) of observed values, with underpredictions of 0.3–0.4 m (0.9–2.3 feet) in scrub oak to as much as 0.6 m (2.0 ft) in pitch pine thin–mow plots. Predicted rates of spread were generally within 0.8 m/min (2.6 ft/min). Underpredictions of 1.8 m/min (6 ft/min) occurred in pitch pine thin–mow plots and of 0.7 m/min (2.3 ft/min) in pitch pine control and thin–graze plots. Observed and predicted values varied less in scrub oak plots, where treatments were applied more homogeneously (Patterson et al. 2005). Fuel loadings and depths in the untreated plots in our study were similar to those throughout the forest (Mouw 2002), which suggests that our custom fuel models are widely applicable.

In oak woodland and scrub oak stands, mowing and grazing effectively reduce shrub and slash loads. Unlike grazing (or slash pile burning), mowing does not consume fuels and, in fact, increases fine litter loads. However, the fine fuels created by mowing are

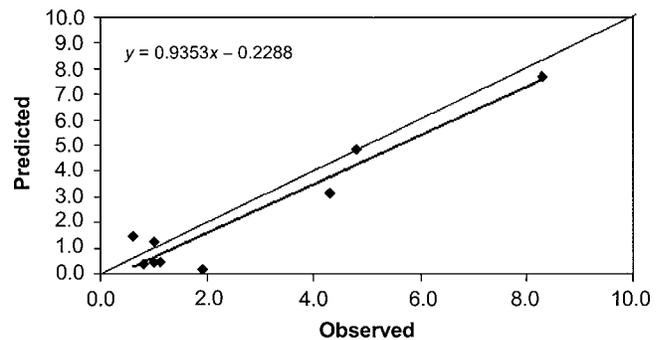


Fig. 7. Predicted versus observed rates of spread (m/min) for prescribed burns in control and fuel treatment plots, Manuel F. Correllus State Forest on Martha's Vineyard, Massachusetts, 2002–2004. The 1:1 diagonal line represents a perfect fit. Observed values for treatment plots are all ≤ 2 m/min.

Table 7. Average (\pm 95% confidence intervals) percent of plot areas burned in April–May 2004, Manuel F. Correllus State Forest, Martha's Vineyard, Massachusetts.

Cover type	Treatment	% burned
Pitch pine	Control	100 \pm 0
	Thin–mow	85 \pm 50
	Thin–graze–pile burn	80 \pm 35
Oak woodland	Control	100 \pm 0
	Mow–graze	85 \pm 85
	Mow	100 \pm 25
Scrub oak	Control	100 \pm 15
	Mow–graze	80 \pm 80
	Mow	100 \pm 10

highly compacted and their flammability is lowered. After initial mowing (necessary to provide palatable stems for sheep) either grazing or mowing can be used to maintain reduced shrub growth and have similar effects on fuels and fire behavior. Grazing was, however, far more expensive than mowing. Costs of the treatments (in 2004 U.S. dollars per hectare) were mowing, \$480; grazing, \$2,160; pile burning, \$480; thinning pitch pine, \$3,600.

Thinning pitch pine stands reduces the potential for crown fires. An active crown fire in a pitch pine stand thinned to 4.5–7 m²/ha (20–30 ft²/acre) requires greater 6-m (20-ft) windspeeds (98 km/h [61 mi/h]) than untreated stands (34 km/h [21 mi/h]; Duveneck and Patterson 2007). Slash from thinning contributes to surface-fire intensity, so treatment of slash is desirable. Mowing following thinning treats slash and shrub fuels simultaneously, but slash fuels can also be piled, covered, and burned in the winter. The costs of mowing and pile burning treatments are comparable. Thinning is not advised for oak woodlands. Because oak leaves are less flammable than pine needles and their canopies reduce surface wind speeds (Mouw 2002), they should be retained.

Without repeated treatments, shrubs—especially scrub oak—resprout vigorously, and fuel loads and depths can be expected to recover within 3–5 y (Nelson 2001). Season of treatment affects recovery rates, with growing-season treatments reducing the vigor of sprouts compared to dormant-season treatments. Because reduced litter production and height growth limit fine fuel loads and fuel bed depths, treatment intervals can be extended if treatments are applied during the growing season. Treatment options may be narrowed, however. Brush cutting and grazing are more practical than burning because summer droughts lead to prolonged smoldering and smoke management problems. Smoke is a particular problem in areas with large summer tourist populations.

CONCLUSIONS

Although our study encompassed only the first two post-treatment years of what should be a long-term study, our initial results documented changes in rare plant abundance, fuel loads, and fire behavior, which suggest that the fuel reduction practices we evaluated are effective both in reducing immediate fire hazard

and improving habitat for rare species. The results of this study have broader implications for managing fuels and rare species habitat in pine–oak forests of the Atlantic Coastal Plain. Thinning pitch pine and treatment of shrub understories in all three cover types not only reduced fuel loads and depths but also increased the abundance of the grassland-associated species that indicate potential rare plant habitat (Clarke and Patterson 2007). Thus, it may be possible to recreate the open, savanna-like woodlands that are believed, by some, to have provided habitat for early successional plant and animal species (e.g., the now-extinct heath hen [*Tympanuchus cupido cupido*]) in precolonial landscapes frequently burned by American Indians (Day 1953). Creation of rare plant habitat may require less effort in scrub oak stands, which have lower duff depths than pitch pine and oak woodlands. But when mowing is applied, it adds a layer of mulch where slash is present and/or shrubs are abundant. Without prescribed fire, mowing may reduce habitat suitability for rare plants. Scrub oak—beneath a scattered canopy of pitch pine or oaks—is also the preferred habitat for at least some of the rare Lepidoptera that this ecosystem supports (Patterson et al. 2005, Haggerty 2006). Restoration and maintenance of open woodlands on the Atlantic Coastal Plain thus meets both fire management and rare species conservation objectives, and our results provide managers with options for accomplishing those objectives.

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PRESCRIBED BURNING REDUCES COMPETITION AND IMPROVES GERMINATION OF TWO RARE PLANTS IN WASHINGTON

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ABSTRACT

We compared the effects of prescribed burning in spring and fall on longsepal wild hollyhock (*Iliamna longisepala*) and Thompson's clover (*Trifolium thompsonii*), two rare native forbs of north-central Washington State. These species occur in dry, scattered ponderosa pine (*Pinus ponderosa*) forests or grasslands where prescribed fire is currently used as one of several tools for forest restoration. We used field experiments conducted over 3 y to explore several aspects of species' response to timing of burning: plant survival, changes in plant size and morphology (including allocation to flowering stems), and changes in population size structure. Fall and spring burns were of comparable severity (proportion of ground surface blackened), but intensity (flame length and maximum temperature) was greater in the fall. Nonetheless, survival of mature *Iliamna* plants was high (82–100%) and did not differ among treatments. In contrast, first-year survival of germinants stimulated by fire was low. Mature *Trifolium* displayed moderate rates of dormancy (3–19%), making determination of survival impossible; however, post-treatment densities were comparable to initial densities, suggesting high survivorship. For both species, plant size (height and crown width), morphology (number of stems and leaves), and reproduction (number of flowering stems) were generally unaffected by burn treatment. Instead, we observed considerable variation in these traits within and among sites and from year to year, suggesting strong controls of local environment and weather on plant performance. For both species, densities of major life stages (seedling, vegetative, and flowering) were also largely unaffected by treatment. However, at one site, fall burning stimulated significantly greater germination of *Iliamna* than did spring burning (although subsequent mortality was high in both treatments). Our results suggest that prescribed fire can have neutral or potentially positive effects on populations of *Iliamna* and *Trifolium*. Timing of burning has little effect on the performance of extant plants, but fall burning can lead to increases in plant density in *Iliamna* by stimulating germination of buried viable seed. For *Trifolium*, frequent or more intense fire may be necessary to reduce competition and maintain suitable habitat conditions for long-term maintenance of populations.

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USING MODELS TO ASSESS FIRE REGIME CONDITIONS AND DEVELOP RESTORATION STRATEGIES IN GRASSLAND SYSTEMS AT LANDSCAPE AND GLOBAL SCALES

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ABSTRACT

Altered fire regimes are a serious threat to biodiversity in almost every major habitat type on earth. Threats to the restoration and maintenance of intact fire regimes (e.g., federal and state fire policies, land use, social values, global plant dispersal, governmental cultures, climate change, etc.) can act at various geographic levels. However, strategies to address these threats differ greatly depending on whether the work is at a landscape or global level. For the purposes of global biodiversity conservation, practitioners and scientists must understand the relationships between local and global ecological processes and conservation strategies, and assess if strategies being applied in one part of a major habitat type can be applied effectively to another. One step toward greater understanding is to compare the similarities and differences in native fire processes and threats within major habitat types. Since 2004, The Nature Conservancy has been working with the USDA Forest Service and U.S. Department of the Interior to quantify reference conditions for every biophysical setting in the United States as part of the LANDFIRE project (www.landfire.gov). The Nature Conservancy is also embarking on an assessment of major habitat types worldwide as part of development of a strategic, global, 10-y goal for biodiversity conservation. Assessment of reference fire regime conditions is a part of this process. We compare similarities and differences between reference condition models developed for 1) grasslands of the midwestern United States as a part of the LANDFIRE project, and 2) all temperate grasslands, savannas, and shrublands at the global scale as a part of The Nature Conservancy's strategic conservation planning. Quantitative state-and-transition models for grasslands of the Midwest were parameterized for the LANDFIRE project using the best available data and local expert knowledge of mean fire frequency, intensity, and ecosystem-level fire effects. Model outputs include the long-term mean percentages of structural-seral stages expected by biophysical setting (i.e., potential natural vegetation type) given a native fire regime. A global-level state-and-transition model for temperate grasslands, savannas, and shrublands was built starting with the LANDFIRE models, given current knowledge about coarse-resolution transitions within this major habitat type. We compare model simulation results for reference conditions at local and global scales for grasslands, savannas, and shrublands, and present results for alternative future conditions at each scale given varying assumptions about native fire regime function and human interactions. Simulation models of ecosystem function at multiple scales are invaluable to the testing of assumptions about ecosystem function, the efficacy of conservation strategies, and their applicability across geographic scales.

Citation: Shlisky, A.J., S. Hickey, and T.B. Bragg. 2007. Using models to assess fire regime conditions and develop restoration strategies in grassland systems at landscape and global scales [abstract]. Page 84 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

ASSESSING MULTIPLE PROCESSES, INCLUDING FIRE, FOR A REGIONAL ASSESSMENT OF GRASSLANDS AND SHRUBLANDS BASED ON NRCS ECOLOGICAL SITES FRAMEWORK

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ABSTRACT

The Nature Conservancy is working with the Bureau of Land Management to assess multiple indicators of ecological condition, including fire regime, across grasslands and shrublands in southern New Mexico. The purpose of the assessment is to identify restoration opportunities and support integrative land management (e.g., fire and range programs). Ecological site descriptions, developed by the USDA Natural Resources Conservation Service (NRCS), are the basis of the assessment. Ecological sites are biophysical units that comprise associations of soils, topography, climate, and potential vegetation. Ecological site descriptions describe reference and non-reference states, processes that maintain these states or drive transitions between them, and potential options for managing states towards desired conditions. The descriptions include advanced state-and-transition models that depict complex ecological dynamics, nonlinear pathways, and ecological thresholds. The role of fire in maintaining reference and non-reference states, and as a possible restoration tool, is described and mapped in relation to multiple interacting ecological processes. In this assessment, ecological sites are identified and mapped from digitized NRCS soil surveys, and reference and non-reference states are mapped by experts with knowledge of specific areas and through remote sensing analysis. Each state is attributed in GIS with ecological processes that influence ecosystem functionality, and with restoration options, based on the ecological site descriptions. The assessment is designed to support statewide and field office-level planning and priority setting. However, since the underlying spatial units, soil survey map units, were developed at fine to medium scale (1:24,000–1:100,000), the assessment is also informative about individual landscapes and watersheds.

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FIRE AND RESOURCE AVAILABILITY INFLUENCE CARBON AND WATER RELATIONS OF THE C₃ SHRUB ROUGHLEAF DOGWOOD IN A MESIC GRASSLAND

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ABSTRACT

A dramatic increase in cover by woody vegetation has been observed in grasslands worldwide, due to independent and interacting global changes. In the C₄-dominated mesic tallgrass prairie of North America, the most proximate factor driving this shift in growth-form dominance is fire exclusion. In 2001, we reintroduced annual fire into a C₄-dominated mesic grassland where it had been excluded for >10 y, and evaluated changes in stem density and ecophysiology of an increasingly abundant C₃ woody plant, roughleaf dogwood (*Cornus drummondii*). Our primary objective was to quantify the impacts of fire-induced aboveground mortality and resource constraints on the persistence of roughleaf dogwood in this ecosystem.

In both years of the study (2001–2002), burned shrubs suffered a complete loss of aboveground biomass due to top-kill from spring fire but persisted via resprouting. A considerable reduction in total growing-season precipitation in 2002, as compared to 2001, revealed that resprouts had lower mortality with reduced soil moisture, were less vulnerable to reductions in soil moisture availability, and maintained higher predawn xylem pressure potentials and rates of photosynthesis than did unburned shrubs. Additionally, enrichment of $\delta^{13}\text{C}_{\text{leaf}}$ in burned shrubs in 2002 was indicative of greater water use efficiency of resprouting stems. During the drought that occurred in the second growing season, mortality of stems within unburned shrub islands increased and was nearly 9 times more than in burned shrub islands. Thus, resprouting shrubs—with their characteristics of reduced leaf tissue, taller stems, and greater stem densities—are efficient in rapidly reoccupying post-fire environments, even with the added stress of below-average precipitation during the growing season. The results of this study suggest that within this grassland, reintroduction of fire will not eliminate roughleaf dogwood in the short term because of its ability to resprout vigorously and persist, even under stressful moisture conditions. Given that North American grasslands are important both economically and ecologically, fire management plans should therefore emphasize frequent fire to prevent shrub establishment.

keywords: C₃ shrubs, carbon isotope ratios, *Cornus drummondii*, fire, Kansas, mesic grasslands, roughleaf dogwood, woody plant encroachment.

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INTRODUCTION

Worldwide, dramatic shifts have occurred in the relative abundances of grass versus woody vegetation, where a growth-form transition to woody dominance has been ascribed to both independent and interacting global changes (Schlesinger et al. 1990, Scholes and Archer 1997, Van Auken 2000, Archer et al. 2001). C₃ woody plant species are native to mesic grasslands of central North America but historically have been maintained as minor components of this ecosystem (Hulbert 1986) dominated by warm-season C₄ grasses that respond positively to frequent fire (Knapp and Seastedt 1998). When the time interval between subsequent fire events is extended (≥ 4 y), shrub cover and abundance may increase rapidly (Briggs et al. 2002, Heisler et al. 2003) while grass productivity declines (Knapp and Seastedt 1986). In extreme cases of fire suppression,

grasslands have become completely dominated by trees and shrubs (Hoch et al. 2002), demonstrating the integral nature of fire in influencing grass–shrub interactions.

The replacement of C₄ grasses by C₃ shrubs requires a transition phase during which small shrubs compete directly with grasses for resources. While many shrub species resprout in response to topkill from fire (Knapp 1986, Hodgkinson 1998, Hoffmann 1998, Brown and Archer 1999), new shoots produced following fire events confront a suite of biotic and abiotic conditions that may constrain their ability to gain carbon. Given their morphology, fire may negatively impact established shrubs 1) directly through loss of woody support tissues and destruction of aboveground meristems, or 2) indirectly via increased competition for soil resources (nutrients and water) from fire-stimulated grasses. Nitrogen (N) is the nutrient most likely to limit productivity in this system (Risser and Parton 1982); thus, competition between grasses and shrubs

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in mesic grasslands is considered strongest for this resource (Kochy and Wilson 2000). Additionally, annual fire reduces available N in the soil due to consistently high levels of plant production (Blair et al. 1998). Because fires in this region occur most often in spring (March–May), the combined effect of removing accumulated plant litter and low albedo of blackened soils contributes to rapid warming and subsequent drying of the soil (Knapp and Seastedt 1986). Such conditions may negatively impact plant–water relations, further constrain carbon gain, and reduce the ability of resprouting shrubs to recover and reoccupy sites. We expected that shrubs in this transition stage (young and recently established) would show clearest evidence of the adverse effects of fire and reduced resource availability, given a less developed root system and smaller carbohydrate stores.

From a management perspective, understanding the mechanisms that constrain or facilitate growth of shrubs during this transition stage is essential. We refer to this as a “transition stage” because, from a plant community perspective, young shrubs in this stage exist as 40–60 clumped stems (>0.8 m in height) within a matrix of C_4 grasses and C_3 forbs (circa 5–8 m² in area). While many shrub islands may be spread across the landscape, herbaceous vegetation biomass remains high and represents fuels effective at carrying fire. In contrast, advanced stages of shrub growth and expansion in this ecosystem exhibit limited herbaceous biomass (fuels) in and around shrub islands that fail to support fire. Thus, introduction of annual fire during the transition phase may reduce or eliminate shrubs. However, when the fire return interval is extended to >4 y, shrubs rapidly establish and increase in cover (Heisler et al. 2003).

While Lett and Knapp (2003) demonstrated that physical removal of large shrub islands promotes a return to a more grass-dominated ecosystem, the time required for complete restoration is uncertain and the process is labor intensive. Such a landscape in which C_4 grasses and mature C_3 shrubs are codominant has been characterized as an alternative stable state that is biotically maintained by shrub-induced low light levels that eliminate understory grasses (Briggs et al. 2005). This evidence suggests that research focusing on the initial years post-establishment, when shrubs coexist with grasses, may be the most effective time to intervene.

The primary goal of this research was to reintroduce annual fire into a landscape in which it had been excluded for >10 y to assess whether successive fire events would reduce or eliminate recently established shrubs in this ecosystem. We selected roughleaf dogwood (*Cornus drummondii*) as a focal species because it is one of the most abundant shrubs in this grassland and tends to form monospecific stands (or “islands”) as it matures in the absence of frequent fire. Given the difference in total growing-season precipitation between the two years of this study, we were able to quantify the response of both burned and unburned shrubs to “average” (2001) and stressful (2002) below-average moisture conditions.

We tested the following hypotheses: 1) leaf-level photosynthetic rates of resprouts, as compared to unburned shrubs, would be reduced; 2) water limitations in burned shrubs would be greater, with resprouts having lower predawn xylem pressure potentials (XPPs) and greater water use efficiency (WUE) than preexisting stems; and 3) N additions would result in elevated rates of photosynthesis in both burned and unburned shrubs as compared to their respective unfertilized counterparts, suggesting competition with grasses for plant-available N in the soil. To test these hypotheses, we utilized both instantaneous and integrative measurements of plant carbon and water relations (leaf-level photosynthetic rates, predawn XPPs, and carbon isotope ratios) and estimated stem densities of burned and unburned shrubs in a factorial experiment that included fire and N addition during the 2001–2002 growing seasons.

STUDY AREA

This study was conducted during the 2001 and 2002 growing seasons at the Konza Prairie Biological Station (KPBS), a 3,487-ha tract of unplowed native tallgrass prairie located approximately 15 km south of Manhattan, Kansas (39°05′N, 96°35′W). KPBS is located within the Flint Hills and exists as the largest continuous tract (1.6 million ha) of unplowed tallgrass prairie in North America. KPBS is characterized by a stream-dissected landscape and rolling hills eroded from chert-bearing limestone and shale of Permian age. Elevation ranges from 320 m in the lowlands to 440 m in uplands and soil (Oviatt 1998). Soil depth varies with topographic position, with shallow, rocky upland soils and considerably deeper lowland soils that may extend to >1 m. In 1981, KPBS was incorporated into the National Science Foundation’s Long-Term Ecological Research network. The location of this study was an experimental unit that has been designated as unburned and ungrazed since 1981; however, a wildfire occurred in 1991, resulting in a 10-y post-fire recovery.

The plant community of KPBS was dominated by C_4 perennial grasses, such as big bluestem (*Andropogon gerardii*) and Indiangrass (*Sorghastrum nutans*), but the presence of several hundred native forb species (non-graminoid herbs) greatly increased plant community diversity (Freeman 1998). C_3 shrubs such as roughleaf dogwood and smooth sumac (*Rhus glabra*) were native to this ecosystem and became increasingly abundant in areas in which fire occurred less frequently (Bragg and Hulbert 1976, Heisler et al. 2003).

The climate of KPBS was characterized by a high degree of interannual variability in precipitation. Mean annual precipitation (100-y average) was 835 mm, 75% of which fell during the growing season (April–September; approximately 630 mm). Precipitation during the 2001 growing season was 720 mm and above average; however, only 510 mm of rain fell during the 2002 growing season.

METHODS

Experimental Treatments

In the spring of 2001, 32 shrub islands of rough-leaf dogwood were selected from an experimental unit within KPBS (approximately 37 ha in area) in which fire had been excluded for >10 y. Because we were interested in the transition phase that precedes dominance by this species, we selected individuals that were 3–5 y old (as determined by counting annual growth rings) and currently nested within a matrix of native grasses and forbs. All islands were located in upland areas with similar topography and soil type. The experimental design was based on a factorial manipulation of fire and nitrogen and therefore included four treatments described below. All fire treatments are described in reference to the entire study site, which was burned on 24 April 2001 and 29 April 2002. Each shrub island was randomly assigned to one of the following four treatments ($n = 8$ shrub islands/treatment type):

1) *Unburned islands*.—Shrub islands were protected from spring burning (24 April 2001; 29 April 2002) of the entire watershed by burning a mowed 1-m buffer around the perimeter of each island. Islands were then surrounded with large strips of sheet metal to protect them from potential scorch damage. In this way, we were able to burn the area immediately adjacent to each island without affecting the interior.

2) *Unburned + nitrogen islands*.—Shrub islands were protected as described above, and nitrogen (10 g N/m^2), in the form of NH_4NO_3 , was applied to each island on 15 June, 2001 and 2002. This amount has been shown to stimulate aboveground net primary productivity (ANPP) in annually burned tallgrass prairie (Seastedt et al. 1991, Knapp and Seastedt 1998).

3) *Burned islands*.—Shrub islands were burned along with surrounding vegetation when the entire watershed was burned.

4) *Burned + nitrogen islands*.—Shrub islands were burned as above treatment 3 and nitrogen was applied as above treatment 2.

Population Parameters

Prior to the first fire in April 2001, stem density was estimated within each shrub island by counting the total number of stems (stems/m^2). In May 2001, all stems in unburned islands were tagged and given a unique identification number. In order to document additional stem production (“new stems”) or mortality from May to September, shrub islands were revisited at approximately 2-week intervals for a total of six tagging events. A similar protocol was followed in 2002; however, only stems produced in that year required a tag, as tags from the 2001 growing season remained attached to stems. In burned shrub islands, the first visit immediately following the April 2001 fire revealed that 100% aboveground mortality had occurred and no live stems were present to be tagged. Similar to the protocol followed in the unburned shrub islands, burned shrub islands were revisited at approx-

imately 2-week intervals throughout the growing season. During each visit, tags bearing unique identification numbers were applied to all resprouts and new stems. We characterized “resprouts” as shoots produced at the base of charred stems (basal resprouts) and “new stems” as shoots produced from underground rhizomes.

The height and basal diameter of individual stems were measured in a subset ($n = 5$ per treatment) of all shrub islands at the end of the growing season in both 2001 and 2002. Given the variability in stem heights within islands, especially those with multiple stem cohorts, we characterized stem height according to the tallest 10 preexisting stems (stems initiated prior to 2001) as well as the tallest new stems produced in a given year (2001 or 2002, respectively).

Field Measurements of Carbon and Water Relations

Gas exchange (net photosynthesis, A_{net}) of rough-leaf dogwood was measured between 1100 and 1300 hours CST under high light conditions with an LI-6200 portable photosynthesis system (Li-Cor, Lincoln, NE) equipped with a 0.25-L chamber. Measurements were taken at approximately 15-d intervals from June to August and were conducted on individual leaves within five randomly selected shrub islands among the eight replicates per treatment. For each shrub island, two single attached upper canopy leaves were measured and then collected to determine leaf area, using an LI-3100 leaf area meter (Li-Cor, Lincoln, NE). On the same days, water availability was estimated by measuring predawn XPP in the field on either resprouts from burned shrubs or preexisting stems in unburned shrubs between 0530 and 0630 hours CDT using a Scholander-type pressure chamber (PMS Instruments, Corvallis, OR).

Nitrogen and Isotopic ($\delta^{13}\text{C}$) Analyses of Leaf Tissue

Leaf tissue from roughleaf dogwood and big bluestem (the dominant C_4 grass on KPBS) was collected on 14 June (early season, pre-N addition) and 15 July (mid-season, post-N addition) in both 2001 and 2002 in order to quantify leaf N. Tissue was sampled from all shrub islands. Leaves were dried at 60°C for approximately 1 week, ground and analyzed for total N content (% dry weight) by combustion and gas chromatography on a Carlo Erba NA 1500 analyzer (Carlo Erba, Milan, Italy). It is important to note that the response of big bluestem was not a focus of this experiment; however, we were interested in determining whether N was a limiting resource for which grasses and shrubs were competing. Elevated levels of N in tissue of big bluestem and/or roughleaf dogwood in response to N additions would suggest N limitation and subsequently competition for this soil resource.

To quantify WUE, a 6.5- to 7.0-mg subsample of roughleaf dogwood leaf tissue was used for carbon isotope analyses. $\delta^{13}\text{C}_{\text{leaf}}$ was measured via combustion using a ThermoFinnigan Delta Plus mass spectrometer (ThermoFinnigan, Bremen, Germany) equipped with an elemental analyzer and continuous flow interface.

Table 1. Population characteristics of burned and unburned roughleaf dogwood shrub islands, Konza Prairie Biological Station, Kansas, in 2001–2002. Values presented are treatment means \pm 1 SE. Significant differences between treatments for a given year are indicated by different lowercase letters.

Characteristic	Pre-fire	2001		2002	
		Unburned	Burned	Unburned	Burned
Stem density (no./m ²)	4.16 \pm 0.31	9.14 \pm 0.78a	16.22 \pm 1.00b	13.85 \pm 1.35a	23.13 \pm 1.85b
Stem height (m)					
Preexisting		0.72 \pm 0.05a	NA ^a	0.77 \pm 0.05a	NA
Resprouts–new stems		0.32 \pm 0.02b	0.45 \pm 0.02c	0.33 \pm 0.03b	0.38 \pm 0.01b

^a In 2001–2002, 100% mortality occurred to preexisting stems in burned shrub islands; consequently, there are no data available (NA) for this stem class.

All carbon isotopes are expressed in delta notation (δ) and calculated according to a within-laboratory standard. Precision of measurements is 0.1–0.2‰.

Statistical Analyses

Data were analyzed using a 2-factor analysis of variance (ANOVA; Proc GLM) with fire (burned or unburned) and nitrogen (control or addition) as main effects (for a total of four treatments). To calculate seasonal means, measurements of net photosynthesis and XPP were analyzed collectively from all days within a growing season. For leaf tissue N analyses, a repeated-measures ANOVA, using initial conditions as covariates, was used to evaluate time and treatment effects as well as interactions. In order to improve normality, an arcsine square-root transformation was conducted on all proportional data. Values presented are means \pm 1 SE, with back-transformed means where appropriate. Duncan's Test was used to contrast group means after significant ANOVA results. All analyses were conducted in SAS 8.1 (SAS Institute, Cary, NC) and the level of significance for all tests was $P < 0.05$, unless otherwise indicated.

RESULTS

Population Parameters

Prior to the first fire in 2001, stem density in shrub islands was 4.16 ± 0.31 stems/m² and similar among all roughleaf dogwood shrub islands. By the end of the first growing season, the density of stems within burned islands increased approximately 4-fold (16.22 ± 1.00 stems/m²) and was significantly greater ($P < 0.05$) than in unburned islands, where a 2-fold increase was observed (9.14 ± 0.78 stems/m²; Table 1). Following fire in 2002, stem density in burned shrub islands further increased to 23.13 ± 1.85 stems/m², which again exceeded new stem production in unburned shrub islands where stem density had risen to 13.85 ± 1.35 stems/m² by the end of the growing season. In burned shrub islands, new stems allocated greater resources to stem rather than to leaf tissue (see Heisler et al. 2004 for details) and reached an average height of 0.45 ± 0.02 m by the end of the growing season in 2001 (Table 1). Compared to new stems in unburned islands, new stems in burned islands were significantly ($P = 0.01$) taller, but remained shorter in stature than preexisting stems (0.72 ± 0.05 m). In

2002, preexisting stems in unburned islands did not increase significantly in height across the growing season, likely due to their relatively large investment to leaf tissue as compared to new stems (Table 1). Height of new stems in burned and unburned shrub islands did not differ significantly in 2002 ($P = 0.38$; Table 1).

Carbon and Water Relations

Seasonal averages of net photosynthesis for resprouts and unburned stems in shrub islands differed dramatically during the two growing seasons of this study. In 2001, fire was significant as a main effect ($P < 0.01$) and reduced photosynthetic rates of resprouts within burned shrub islands (11.11 ± 0.57 $\mu\text{mol m}^{-2} \text{s}^{-1}$) relative to preexisting stems of unburned shrubs (12.79 ± 0.67 $\mu\text{mol m}^{-2} \text{s}^{-1}$; Figure 1). The addition of N did not significantly affect either treatment group ($P = 0.14$). In 2002, fire was again significant as a main effect ($P < 0.01$) and resulted in higher average photosynthetic rates in resprouts of burned shrubs as compared to unburned shrubs (5.34 ± 0.54 $\mu\text{mol m}^{-2} \text{s}^{-1}$ versus 6.81 ± 0.56 $\mu\text{mol m}^{-2} \text{s}^{-1}$; Figure 1). Similar to results for 2001, no effect of N was observed ($P > 0.05$).

Seasonal averages of XPP demonstrated a consistent pattern in both 2001 and 2002, with resprouts having significantly higher values than unburned shrubs (Figure 1). Fire was significant as a main effect ($P < 0.05$), with resprouts of burned shrubs having higher XPP (-0.86 ± 0.08 MPa) than preexisting stems in unburned shrub islands (-1.21 ± 0.12 MPa). The addition of N did not influence XPP in either year of the study ($P > 0.05$).

N content was elevated in leaf tissue from resprouts in burned shrub islands, indicating that the main effect of fire was significant ($P < 0.05$). In June 2001, leaf tissue N was $2.84 \pm 0.06\%$ in burned shrub islands and was reduced across the duration of the growing season to $1.87 \pm 0.05\%$, as measured in July (Table 2). While a similar reduction was observed in unburned shrub islands, leaf tissue N was significantly less ($P < 0.05$) during both sampling periods ($2.19 \pm 0.09\%$ and $1.72 \pm 0.04\%$, respectively; Table 2). A similar trend was observed for roughleaf dogwood in 2002 and consistent in both years was no effect of N addition on leaf tissue N content ($P > 0.05$).

In order to determine whether the dominant grasses within shrub islands had responded to fertilization, we measured leaf tissue N in big bluestem. While the

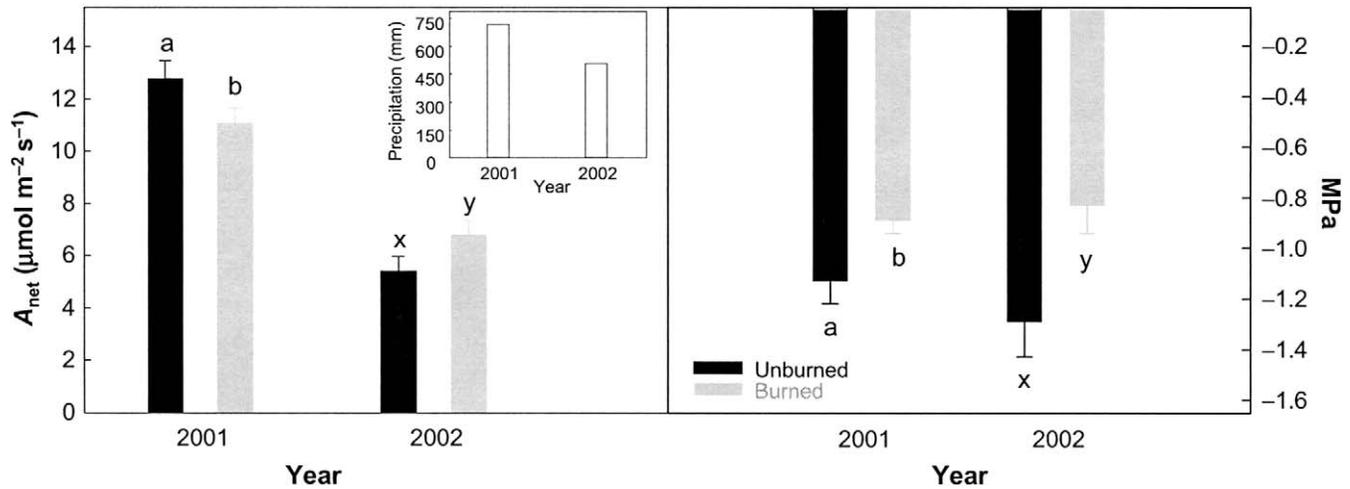


Fig. 1. Seasonal means for net photosynthetic rates (A_{net} ; left), predawn xylem pressure potentials (right), and precipitation (inset) during the 2001–2002 growing seasons for burned and unburned shrub islands of roughleaf dogwood, Konza Prairie Biological Station, Kansas. Vertical bars represent ± 1 SE. Within years and according to measurement type, significant differences ($P < 0.05$) are indicated by different letters.

main effect of fire did not significantly influence leaf tissue N content in big bluestem ($P > 0.05$), in 2001 the addition of N resulted in an increase in leaf tissue N content in fertilized islands as compared to controls ($1.44 \pm 0.06\%$ versus $1.26 \pm 0.03\%$; Table 2). There was no effect of fire or N observed in 2002.

In 2001, burned and unburned shrubs had similar carbon isotope ratios and thus WUE over the course of the growing season (Figure 2). In contrast, growing-season precipitation was reduced by approximately 200 mm in the 2002 growing season, and resprouts of burned shrubs were enriched in $\delta^{13}\text{C}$ ($-25.8 \pm 33\%$) as compared to unburned shrubs ($-26.90 \pm 0.25\%$). This enrichment of ^{13}C in the leaf tissue of burned shrubs indicates greater WUE (Figure 2).

Stem Mortality

An assessment of growing-season stem mortality revealed that in 2001, rates were significantly greater

within unburned shrub islands ($8.45 \pm 1.51\%$) than within burned shrub islands ($2.50 \pm 0.06\%$; $P < 0.01$; Figure 3). While similar rates were observed in 2002 for burned shrub islands ($2.72 \pm 0.67\%$), mortality rates in protected islands increased to $17.58 \pm 2.76\%$.

DISCUSSION

Grasslands are structured by the interaction of fire, climate variability, and herbivory (Cowles 1928), and tallgrass prairie is no exception. Historically, this combination of factors has mediated a tension zone between C_4 grass and C_3 woody species, resulting in an ecosystem that is dominated by herbaceous plants. During the last century, however, reductions in the intensity and frequency of fire have been associated with an increase in woody vegetation abundance. Roughleaf dogwood has proven to be a particularly aggressive species, as mature shrub islands completely exclude

Table 2. Leaf tissue nitrogen (N; %) for roughleaf dogwood and big bluestem within shrub islands, Konza Prairie Biological Station, Kansas, in 2001 and 2002. Samples were taken in June (pre-N addition) and July (approximately 1 mo post-N addition). Values shown are means ± 1 SE. Lowercase letters indicate significant differences ($P < 0.05$) for main effects both within and between time periods.

Year	Species	Main effect	Treatment	Month	
				Jun	Jul
2001	Roughleaf dogwood	Fire	Burned	$2.84 \pm 0.06\text{a}$	$1.87 \pm 0.05\text{c}$
			Unburned	$2.19 \pm 0.09\text{b}$	$1.72 \pm 0.04\text{d}$
		Nitrogen	Control	$2.57 \pm 0.02\text{a}$	$1.82 \pm 0.05\text{b}$
	Big bluestem	Fire	Burned	$2.48 \pm 0.11\text{a}$	$1.78 \pm 0.06\text{b}$
			Unburned	$1.57 \pm 0.04\text{a}$	$1.37 \pm 0.06\text{b}$
		Nitrogen	Control	$1.55 \pm 0.05\text{a}$	$1.33 \pm 0.03\text{b}$
2002	Roughleaf dogwood	Fire	Burned	$1.53 \pm 0.05\text{a}$	$1.26 \pm 0.03\text{b}$
			Unburned	$1.59 \pm 0.04\text{a}$	$1.44 \pm 0.06\text{c}$
		Nitrogen	Control	$2.60 \pm 0.06\text{a}$	$1.60 \pm 0.04\text{c}$
	Big bluestem	Fire	Burned	$2.31 \pm 0.05\text{b}$	$1.47 \pm 0.03\text{d}$
			Unburned	$2.50 \pm 0.06\text{a}$	$1.51 \pm 0.04\text{b}$
		Nitrogen	Control	$2.38 \pm 0.06\text{a}$	$1.55 \pm 0.03\text{b}$
		Fire	Burned	$2.18 \pm 0.04\text{a}$	$1.09 \pm 0.03\text{b}$
			Unburned	$2.04 \pm 0.05\text{a}$	$1.14 \pm 0.03\text{b}$
		Nitrogen	Control	$2.11 \pm 0.04\text{a}$	$1.13 \pm 0.03\text{b}$
		Addition	$2.09 \pm 0.06\text{a}$	$1.11 \pm 0.03\text{b}$	

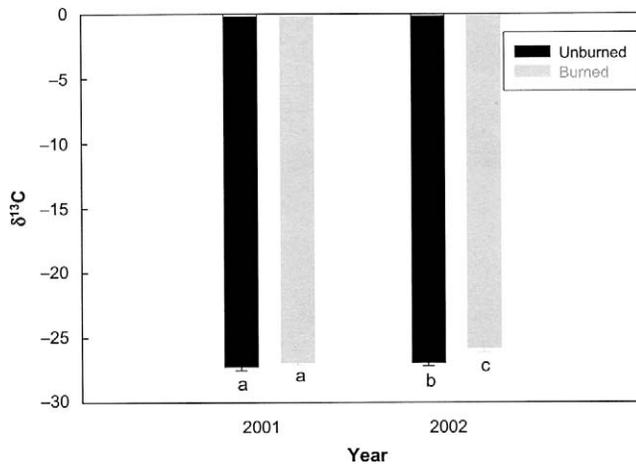


Fig. 2. Carbon isotope signatures for burned and unburned shrubs (roughleaf dogwood) from the 2001 and 2002 growing seasons, Konza Prairie Biological Station, Kansas. $\delta^{13}\text{C}$ values reflect carbon isotope signatures from leaf tissue collected on 15 July, 2001 and 2002. Treatment means ± 1 SE are reported. Means with different letters are significantly different at $P < 0.05$.

understory vegetation by reducing light levels (Lett and Knapp 2003). Mature shrub islands are characterized by >400 dense stems (circa 35 m^2 area), which are in many cases several meters tall, and fire is no longer an effective management tool for reducing shrub cover. Our objective, therefore, was to reintroduce fire into a transitional landscape and document the influence of fire and resource availability on stem densities, leaf-level gas exchange, and water relations.

During this 2-y study, recurrent fire, and its interaction with moisture availability, had significant effects on population characteristics and the ecophysiology of roughleaf dogwood. A high degree of inter-annual variability in precipitation is characteristic of tallgrass prairie (Knapp and Seastedt 1998). In the time frame of this study, considerable differences in growing-season precipitation from 2001 (above average; 720 mm) to 2002 (below average; 510 mm) allowed us to evaluate the response of unburned shrub islands to changes in moisture availability as well as the interaction of fire and short-term drought in resprouts of burned shrub islands. As hypothesized, photosynthetic rates of resprouts from burned shrub islands were lower, on average, than those from unburned shrubs; however, this response was constrained to the 2001 growing season, which was characterized by greater moisture availability. In 2002, improved water relations and greater WUE of resprouts enabled them to better withstand drought conditions, and contributed to greater carbon gain over the long term, via higher rates of photosynthesis. Soil moisture is generally lower in burned as compared to unburned prairie (Hulbert 1969, Rice and Parenti 1978), primarily due to the absence of the litter layer. In unburned shrub islands, the litter layer reduces the amount of light incident to the soil surface and subsequently evaporative demand. This suggests that resprouts in burned shrub islands potentially confront greater moisture constraints. However, morphological characteristics of resprouts versus

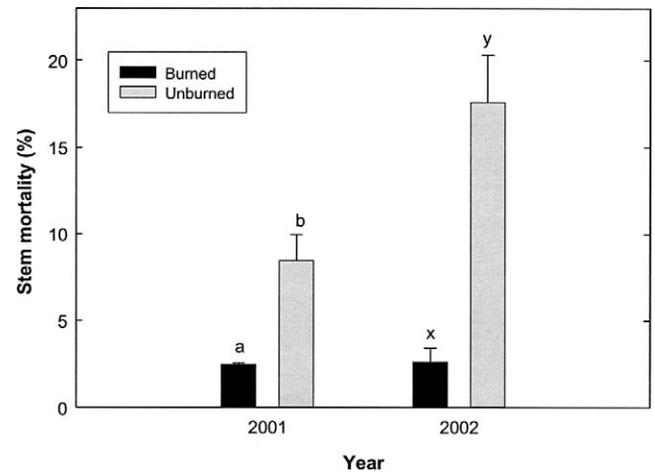


Fig. 3. Mortality (%) of stems in burned and unburned shrub islands of roughleaf dogwood during the 2001 and 2002 growing seasons, Konza Prairie Biological Station, Kansas. Values represent treatment means ± 1 SE and quantify observed stem mortality post-fire and throughout a growing season. Within each year, means with different letters are significantly different at $P < 0.05$.

preexisting stems appear to be more important than soil moisture conditions in determining plant-water relations. While fire results in 100% aboveground mortality, the extensive preexisting root systems of resprouting vegetation together with reduced shoot mass results in greater water availability to growing resprouts (Oechel and Hastings 1983). Furthermore, allocation of biomass to stem tissue rather than to leaf tissue (Heisler et al. 2004) likely reduces whole-plant transpiration, resulting in reduced water stress. Resprouts were less sensitive to significant reductions in growing-season precipitation, as compared to unburned shrubs, which experienced a higher rate of stem mortality (approximately 20%) in 2002. Clearly, short-term drought did not intensify the effects of fire on roughleaf dogwood, in terms of the ability of this species to persist in this grassland.

Following fire, the response of this shrub is to resprout and initiate a new cohort of rapidly growing stems. By the end of the first growing season, resprouts had reached an average height of 0.45 ± 0.02 m and were significantly greater in stature than new stems in unburned islands (0.32 ± 0.02 m). Additionally, stem density had increased nearly 4-fold, with rates of mortality of new stems being approximately 3%. While total ANPP in burned shrub islands is reduced relative to islands that remain unburned (Heisler et al. 2004), persistence of roughleaf dogwood is facilitated by the increase in stem density and height of post-fire resprouts, which enables them to display their leaves in abundant light.

Contrary to our hypothesis that the addition of nitrogen would result in elevated rates of photosynthesis, we could detect no evidence of N limitation in this study. Prior to the initiation of this experiment, fire had been excluded from this site since 1991. Blair (1997) observed that in the absence of fire, a reduction in plant productivity and uptake of soil N results in

the accumulation of this resource. Thus, when N additions accompany fire in long-unburned areas, the production response, if any, is limited. This explains the absence of a response to N addition in 2001. In 2002, a reduction in growing-season precipitation may have resulted in water being a much greater limiting resource rather than soil N. Although resprouts had greater tissue N levels, it was not associated with an increase in photosynthetic rate, suggesting that N was not limiting photosynthesis in roughleaf dogwood.

In summary, short-term reductions in growing-season precipitation, and therefore moisture availability, had a strong impact on seasonal photosynthetic rates and carbon gain of roughleaf dogwood. During a year of above-average precipitation, unburned shrubs had higher rates of photosynthesis as compared to resprouts of burned vegetation. Conversely, when growing-season precipitation was reduced to below average, unburned shrubs experienced greater moisture stress and a reduction in photosynthetic rate as well as elevated rates of stem mortality. While resprouts in burned shrub islands were also impacted by drought, they were able to sustain higher rates of photosynthesis and experienced no increase in stem mortality during the growing season. During dry periods, a well-developed root system and reduced leaf area ameliorate conditions of moisture limitation in resprouts, thereby facilitating persistence of roughleaf dogwood. We present these data with the caveat that responses of roughleaf dogwood are limited to the early stages of restoring annual fire to this ecosystem. We could find no other published studies describing responses of shrubs to an annual fire regime in the long term. While annually burned mesic grasslands are generally devoid of woody vegetation, few data exist to suggest that this is due to the gradual elimination of shrubs. Rather, annual fire instead appears to have prevented shrubs from establishing in C_4 -dominated grasslands. Indeed, further studies are needed of how the functional responses of this shrub are affected by repeated cycles of fire and drought, particularly given that climate models predict extreme weather in the future.

MANAGEMENT IMPLICATIONS

The expansion of woody vegetation in grass-dominated ecosystems is a global phenomenon. Fire suppression is the most cited cause of shrub expansion in mesic grasslands, and the long-term effect of eliminating fire from this ecosystem can be the complete conversion of grassland to shrubland. While long-term data suggest that fire remains effective in preventing shrub establishment, it is currently unclear as to whether management strategies that prescribe frequent fire will be effective in reducing or eliminating shrubs from mesic grassland. Our study results suggest that in the short term, fire does not reduce stem densities within shrub islands and that resprouting shrubs possess morphological attributes that enable them to confront moisture constraints that are typical of grasslands. It is our goal to continue monitoring the re-

sponse of roughleaf dogwood to annual fire in order to determine the long-term efficacy of restoring frequent fire to shrub-invaded mesic grasslands. Given the existing data, management plans should emphasize frequent (<4-y fire return intervals) fire in an effort to prevent shrub establishment.

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IMPACT OF FIRE ON SOIL RESOURCE PATTERNS IN A NORTHERN CALIFORNIA MONTANE ECOSYSTEM

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ABSTRACT

The montane ecosystems of the Cascade Range have been subjected to repeated manipulation and active fire suppression for more than a century. This has resulted in changes in community structure that contribute to increased wildfire hazard and severity. Ongoing efforts to return these ecosystems to a state with intrinsic low fuel loads have received substantial attention in recent years; however, many ecological questions remain unanswered. This study addresses belowground impacts of restorative treatments. We report proximate effects of the application of prescribed fire (burn-only) and the combination of fire and mechanical thinning (thin + burn) on soil chemical and microbial parameters in treatment units of 10 ha each in the Klamath National Forest of northern California. Soil organic carbon (C) decreased and C:N (carbon:nitrogen) ratio increased as a result of fire in the burn-only treatment; however, no significant changes from pre-fire to post-fire were observed for thin + burn treatment plots. N mineralization rates did not change as a result of fire in either burn-only or thin + burn plots. Nitrification rates decreased as a result of fire in thin + burn units, but did not change significantly in the burn-only treatment. Total inorganic N increased from pre-fire to post-fire, with and without thinning. Activity of acid phosphatase, an indicator of overall microbial activity, was reduced by fire, both with and without thinning, whereas activity of chitinase was reduced by fire in thin + burn plots only. There were no significant changes in phenol oxidase activity as a result of fire, with or without thinning. This study demonstrates that fire has short-term effects on soil ecological properties, e.g., soil organic matter, N turnover and availability, and microbial function, and that the combination of mechanical thinning and fire may have different effects from fire alone.

keywords: California, thinning, mixed conifer, prescribed fire, soil enzyme activity, soil resources.

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INTRODUCTION

Fire has been an important process in determining vegetation structure and composition of ecosystems throughout North American history. In western North America pre-European settlement ponderosa pine (*Pinus ponderosa*) (nomenclature follows Hickman 1993) forests experienced fire return intervals as frequent as 2 to 15 y (Weaver 1951, Agee 1993). These fires resulted from both natural causes, such as lightning strikes or volcanic activity, and anthropogenic ignitions, such as seasonal burns set by Native American peoples (Weaver 1951, Agee 1993). The historic pattern of frequent, low-intensity fires limited understory growth and selected for resistant, thick-barked, shade-intolerant tree species such as ponderosa pine. Reports of presettlement forests describe open, park-like stands of trees (Laudenslayer and Darr 1990, Covington and Moore 1994), and detailed dendrochronological anal-

ysis of forests dominated by ponderosa pine and Jeffrey pine (*P. jeffreyi*) in the Cascade Range and Sierra Nevada of California and Oregon profile a pine-dominated forest that was originally moderately open, uneven-aged, large-tree dominated, and shaped by frequent, low-intensity fires (Agee 1993, Skinner and Chang 1996, Taylor 2000).

Today's mixed-conifer forests of the Southern Cascades differ substantially from historic conditions in several characteristics, including increased stand density, altered species composition, increased presence of disease, and nutrient cycling, among others (Agee 1993, Skinner and Chang 1996, Taylor 2000). Grazing, logging, and aggressive fire-suppression policies have contributed to these post-European settlement changes in stand composition and structure (Weaver 1951, Agee 1993). Forests that were historically dominated by a low density of large pines now include large numbers of white fir (*Abies concolor*) and other shade-tolerant species, and stem density in these stands has increased dramatically since European settlement. The

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combination of fire suppression and higher stem density has led to the accumulation of both surface and vertical fuels such that conditions are now conducive to the development of fires with intensity far greater than the historical condition. Large, catastrophic wildfires in California during the summers of 1977, 1987, 1990, 1992, 1999, 2001, and 2003 provide stark evidence of the potential of existing forest conditions for extensive and severe fires.

The problem of hazardous fuel conditions in forests has received substantial attention in recent years as land managers attempt to reduce the threat of catastrophic fires. Reduction of fuel loads, especially in areas of wildland–urban interface, has been conducted primarily via mechanical treatment (usually thinning from below); however, prescribed burning or combinations of thinning and burning are also used. Although such stand manipulations may be able to return forests to approximate historic structural conditions, many questions about the ecological consequences of these fire surrogate treatments remain unanswered (Sierra Nevada Ecosystem Project 1996). For example, the question of the effects of structural and functional manipulations on soil fertility, microbial ecology, and nutrient cycling has only recently begun to attract scientific attention (e.g., DeLuca and Zouhar 2000).

This study details one portion of the National Fire and Fire Surrogate Network study (www.fs.fed.us/ffs), which is designed to evaluate the efficacy of alternative management strategies for the mitigation of the current wildfire hazard and for the improvement of forest ecosystem sustainability and health on a national scale. Our study site in the southern Cascades Range of northern California is one of 13 in which a common experimental design is being employed to determine whether prescribed fire, mechanical treatment (usually thinning from below), or a combination of the two could best 1) minimize the hazard of catastrophic fire and 2) accelerate and sustain the development of a large-tree-dominated, late-successional pine forest similar to that originally created and sustained by the historical fire regime. Within that larger experimental context, we present here the results of a study designed to determine the belowground impacts of prescribed fire, both alone and in combination with thinning.

STUDY AREA

This study took place in the Goosenest Adaptive Management Area (GAMA) of the Klamath National Forest in Siskiyou County, California (lat 41°35'N, long 121°53'W). The forests of GAMA were logged between 1900 and 1920. The gentle, dissected landscape of GAMA was the result of recent volcanic activity. Slopes were generally <10% but could locally be >50%, with elevation ranging from 1,500 to 2,000 m. White fir and ponderosa pine were dominant in the forest canopy, with sugar pine (*P. lambertiana*), incense cedar (*Calocedrus decurrens*), Shasta red fir (*Abies magnifica* var. *shastensis*), and Sierra lodgepole pine (*P. contorta* var. *murrayana*) common. Shrubs in-

cluded ceanothus (*Ceanothus* spp.), manzanita (*Arctostaphylos* spp.), antelope bitterbrush (*Purshia tridentata*), curl-leaf mountain mahogany (*Cercocarpus ledifolius*), rabbitbrush (*Chrysothamnus* spp.), and sagebrush (*Artemisia* spp.) (USFS 1996).

The soils of the experimental area were the Belzar–Wintoner complex of inceptisols mixed with lesser cover of alfisols (USFS 1982). The Belzar series consists of loamy-skeletal, mixed, frigid Andic Xerochrepts. The Wintoner series, pumice overburden phase, consists of fine-loamy, mixed, frigid Ultic Haploxeralfs. These sand and sandy loam soils drain rapidly and have relatively low water-holding capacity. The climate was Mediterranean-type, and the study site received most of the 25- to 100-cm annual precipitation as winter snowfall (USFS 1996).

METHODS

Study Design and Field Methods

The base Fire and Fire Surrogate (FFS) study design is a completely randomized design consisting of four treatments each replicated three times, with pre-treatment sampling to be done in 2000 and treatments implemented in 2001. The intended location of the Southern Cascades FFS study site in the Klamath National Forest became unavailable in 2000, too late for us to restart and complete the permitting process for an entirely new location in time to meet the implementation schedule. In order not to lose the opportunity to include the Southern Cascades in the national FFS study, we decided to overlay the treatment units of this study on preexisting research plots of the Little Horse Peak Interdisciplinary Study (LHPIS). The LHPIS was established in 1998 to accelerate the development of late-successional characteristics of east-side pine forests (Zack et al. 1999, Ritchie 2005). Several of the LHPIS treatments paralleled the base FFS design closely. The combination of mechanical treatment (i.e., thinning from below) and prescribed fire (hereafter thin + burn) was replicated in five experimental units for the LHPIS study, three of which were randomly selected in 2000 for this FFS study. In contrast, the LHPIS did not include a treatment that paralleled the FFS burn-only treatment. Treatment units for FFS burn-only treatments (hereafter burn-only) were interspersed among the Little Horse Peak in 2001. Although vegetation and fuels data were taken in 1998, prior to treatment of the LHPIS units, no pre-treatment soil sampling was done.

The thinning of the thin + burn units involved thinning from below combined with selection cutting that focused on removing shade-tolerant species such as white fir. An average of 34% of the basal area was removed as mean basal area was reduced from 36.5 m²/ha to 24.1 m²/ha. Processing followed standard harvesting procedure for forests in this region. Whole trees were transported to central processing landings where limbs and tops were removed from larger trees and logs cut to appropriate length for hauling to processing plants. Limbs, tops, and small trees were

Table 1. Fire weather and behavior characteristics for prescribed fires in the burn-only and thin + burn treatment areas of the Goosenest Adaptive Management Area, northern California. Means \pm standard deviations are given. Data are from USDA Forest Service, Pacific Southwest Research Station, Redding, California.

Parameter	Treatment	
	Burn-only	Thin + burn
Air temperature ($^{\circ}$ C)	6 \pm 5	12 \pm 5
Relative humidity (%)	24 \pm 13	32 \pm 16
Wind speed (km h $^{-1}$)	1.7 \pm 1.3	2.6 \pm 1.6
Flame length (cm)	24 \pm 11	40 \pm 18

chipped at the landings and removed; thus, all the slash generated by the thinning was removed from the units prior to burning. Removal of slash from landings is the usual practice in this region.

Thinning was done during the summers of 1998 and 1999, and was followed by a prescribed fire in the late fall of 2001. Our soil samples in the thin + burn treatment from the summer of 2001 therefore represent pre-fire but post-thinning, and our 2002 samples represent post-burning and thinning. The burn-only treatment consisted of a late-fall prescribed burn in 2002. Stand structure was not modified mechanically prior to burning in the burn-only units. We sampled the burn-only units in 2002 for pre-burn conditions and in 2003 for post-burn conditions. The lack of synchrony in our sampling was an artifact of our having to overlay the FFS study on the preexisting LHPIS design.

Firing techniques used in the prescribed burning were strip-head firing and tree-centered spot ignition (Weatherspoon et al. 1989). Weather conditions and flame lengths are given in Table 1.

Although the full FFS design includes mechanical thinning, prescribed fire, and their combination, in this paper we focus on the issue of how fire, as a single factor, affects belowground ecological components in relation to vegetation and fuel status, i.e., whether the site had been commercially thinned prior to fire or not. As such, it represents an attempt on our part to focus on a specific aspect of the larger FFS design that is of immediate interest to land managers. In addition, we did not include the untreated control units in this study, as the question on which we wished to focus was one of the effects of fire with and without thinning, not the effects of fire and/or thinning relative to an untreated site.

Soil Sampling and Laboratory Analysis

In each 10-ha FFS treatment unit ten 20 \times 50-m (0.1-ha) random, permanent sampling plots were established on a 50-m grid and the corners marked with permanent posts. Four soil sampling points were located at each of the corners of the 0.1-ha plot, and one sampling point was located at the midpoint of each of the long sides of the 0.1-ha plot. In June–August of 2001–2003 soil samples to 10-cm depth were taken from those six points in each sample plot as follows: thin + burn pre-fire 2001 (after thinning was completed), thin + burn post-fire 2002, burn-only pre-fire

2002, and burn-only post-fire 2003. Soil samples on successive dates in each treatment unit were taken within 1.0 m of those from the previous year. All samples were returned to the laboratory under refrigeration. Each sample was passed through a 5-mm sieve to remove stones and root fragments. Sieved, air-dried soil samples were extracted with 0.5 M K₂SO₄ for NO₃⁻, NH₄⁺ (Olsen and Sommers 1982) and analyzed using the microtiter methods of Hamilton and Sims (1995).

Estimation of nitrogen (N) mineralization and nitrification for all study plots was done using aerobic, in situ incubations following the method of Raison et al. (1987). Groups of three 10-cm-deep polyvinyl chloride (PVC) soil cores were taken at each of the six sampling points around each permanent sampling plot ($n = 720$). One core was returned to the lab immediately, whereas the remaining two incubated in situ for 20–30 d. One of the in situ cores was covered with a PVC cap while the other remained open. As the results from capped and uncapped cores were not significantly different, they were pooled for later data analysis. All three cores from a given sampling point were extracted and analyzed for inorganic N as indicated above. Net N mineralization was calculated as the difference in total inorganic N (NO₃⁻ + NH₄⁺) concentration between the initial samples and those that incubated for 20–30 d. Net nitrification was calculated as the difference in NO₃⁻ in the incubated and initial samples. Proportional nitrification was estimated by dividing the net NO₃⁻ accumulation due to nitrification by total amount NH₄⁺ available to be nitrified (initial NH₄⁺ + net N mineralization).

Two additional soil samples for analysis of acid phosphatase, chitinase, and phenol oxidase activity were taken at opposite corners of each permanent sample plot during each sampling year ($n = 240$). Enzyme activities were analyzed using methods developed by Tabatabai (1982), as modified by Sinsabaugh (Sinsabaugh et al. 1993, Sinsabaugh and Findlay 1995). Subsamples of approximately 10 g of fresh soil were suspended in 120 mL of 50 mM NaOAc buffer (pH 5.0) and homogenized by rapid mechanical stirring for 90 s. To minimize sand sedimentation, stirring was continued while aliquots were withdrawn for analysis.

Acid phosphatase (EC 3.1.3.2) and chitinase (EC 3.2.1.14) activities were determined using *p*-nitrophenol (*p*NP)-linked substrates: *p*NP-phosphate for acid phosphatase and *p*NP-glucosaminide for chitinase. Samples were incubated for 1 h (acid phosphatase) or 2 h (chitinase) at 20–22 $^{\circ}$ C with constant mixing. Following incubation, samples were centrifuged at 3,000 \times *g* for 3 min to precipitate particulates. An aliquot of 2.0 mL of the supernatant was transferred to a clean, sterile tube, and 0.1 mL of 1.0 M NaOH was added to halt enzymatic activity and facilitate color development. Prior to spectrophotometric analysis at 410 nm, each sample of the supernatant was diluted with 8.0 mL of distilled, deionized water.

Phenol oxidase (EC 1.14.18.1, 1.10.3.2) activity was measured by oxidation of L-DOPA (L-3,4-dihydroxyphenylalanine) during 1-h incubations at 20–

22°C. Following incubation, samples were centrifuged as above and analyzed at 460 nm without dilution. Parallel oxidations using standard horseradish peroxidase (Sigma Chemical, St. Louis, MO) were used to calculate the L-DOPA extinction coefficient.

Organic carbon (C) and total N were determined by oxidation/fluorescence on a Carlo Erba CN analyzer (Carlo Erba, Milan, Italy) after grinding air-dried soil samples to pass through a 0.32-mm-mesh screen.

Data Analysis

All response variables were either normally distributed or could be transformed to normality with a square root transformation. As the prescribed fires applied to the burn-only and thin + burn treatments occurred in successive growing seasons, this experiment was not a 2 × 2 factorial design with years and fire as main effects. Instead, we treated it as two separate experiments, each with a completely randomized design, and evaluated the effect of fire on the burn-only and thin + burn units as independent one-way analyses of variance (SAS Institute 2004). Statistical significance is reported at *P* = 0.05.

RESULTS

Fire significantly reduced soil organic C in the burn-only treatment (*P* < 0.001) but not in the thin + burn treatment (*P* < 0.188) (Figure 1a). Fire reduced soil organic C in the burn-only treatment by an average of 29%. Similarly, fire significantly affected soil C:N ratio in the burn-only (*P* < 0.001) but not in the thin + burn treatment (*P* < 0.266) (Figure 1b). In the burn-only treatment, soil C:N ratio increased by 18%. Thus, fire significantly reduced both organic matter quantity and quality in units that had not been mechanically thinned, but not in units that had been thinned prior to burning.

In the burn-only units there was no significant effect of fire on either net N mineralization rate (*P* < 0.987) (Figure 2a) or net nitrification (*P* < 0.874) (Figure 2b). In contrast, in the treatment units that were mechanically thinned prior to burning, fire resulted in a trend toward a reduction in net N mineralization (*P* < 0.097) (Figure 2a) and a significant reduction in net nitrification (*P* < 0.029) (Figure 2b). The reduction in net nitrification rate due to burning in the thin + burn units averaged 31%. Total inorganic N (TIN) in the soil solution increased significantly as a result of fire in both the burn-only and the thin + burn treatments (*P* < 0.001 in both) (Figure 2c).

Acid phosphatase activity was significantly reduced by fire in both treatments (*P* < 0.001 burn-only; *P* < 0.013 thin + burn) (Figure 3a). Acid phosphatase activity was reduced by an average of 42% in the burn-only treatment but by an average of 17% in the thin + burn treatment. Chitinase activity was reduced significantly by fire in the thin + burn (*P* < 0.028) but not in the burn-only treatment (*P* < 0.116) (Figure 3b). Phenol oxidase activity was not significantly affected

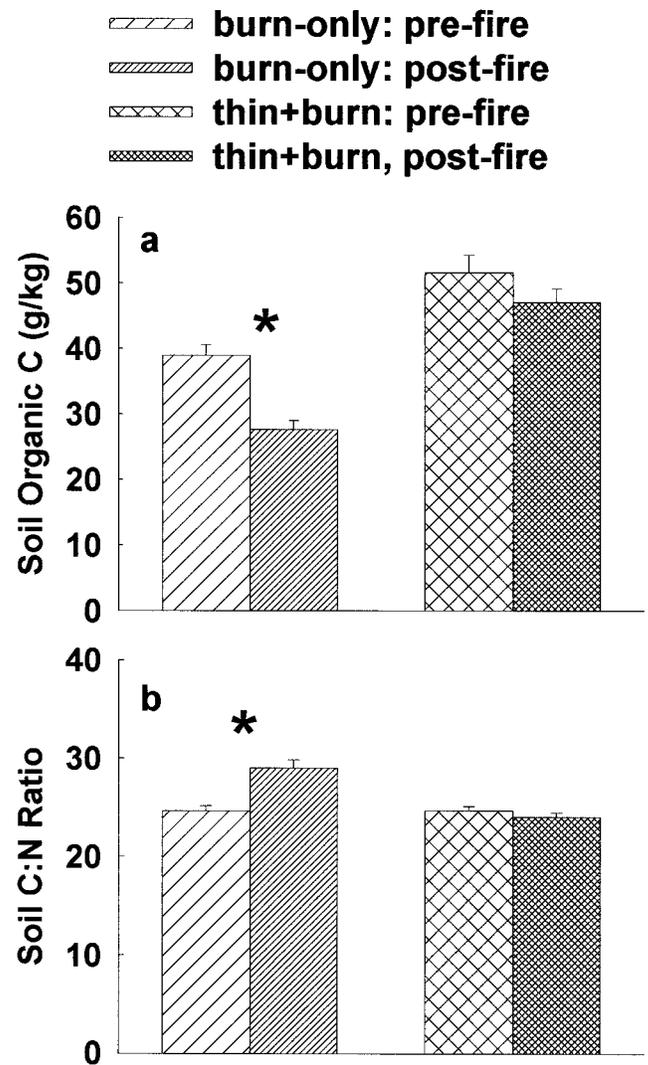


Fig. 1. Soil organic C (a) and C:N ratio (b) before and 1 y after fire (2001 and 2002 for thin + burn treatment; 2002 and 2003 for burn-only treatment) at the Goosenest Adaptive Management Area, northern California. Histogram bars denote means with standard errors of the means; significant differences at *P* = 0.05 are indicated by asterisks.

by fire in either treatment (*P* < 0.103 burn-only; *P* < 0.137 thin + burn) (Figure 3c).

DISCUSSION

The primary objective of this study was to assess proximate belowground responses to prescribed fire, alone and in combination with pre-commercial mechanical thinning. Although the larger National Fire and Fire Surrogate Study, of which this study is a part, was designed to be a long-term analysis of the efficacy of these and other treatments for reducing wildfire hazard and improving ecosystem sustainability, we present here a more focused assessment of the proximate effects of fire in the Southern Cascades FFS study site. We do so both as a benchmark against which to evaluate longer-term ecosystem responses and for compar-

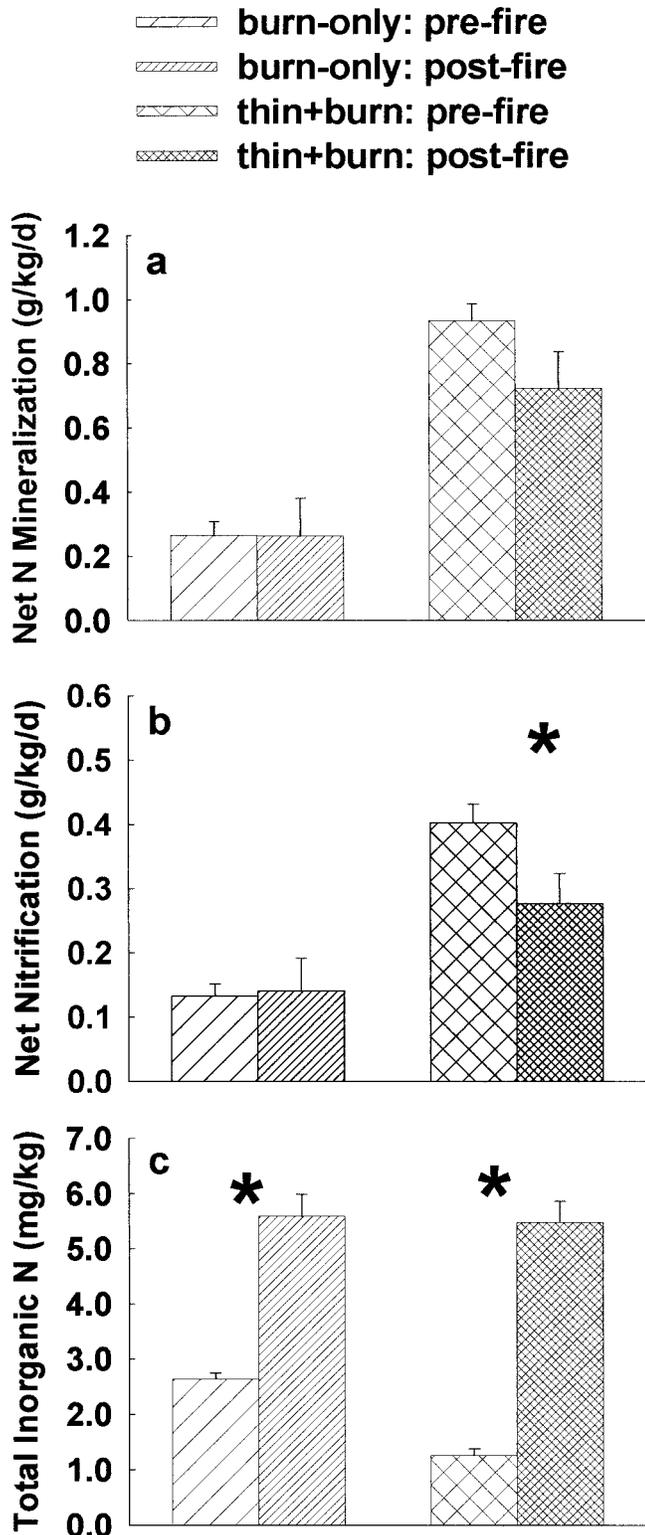


Fig. 2. Net N mineralization rate (a), net nitrification rate (b), and total inorganic N (c) before and 1 y after fire (2001 and 2002 for thin + burn treatment; 2002 and 2003 for burn-only treatment) at the Goosenest Adaptive Management Area, northern California. Histogram bars denote means with standard errors of the means; significant differences at $P = 0.05$ are indicated by asterisks.

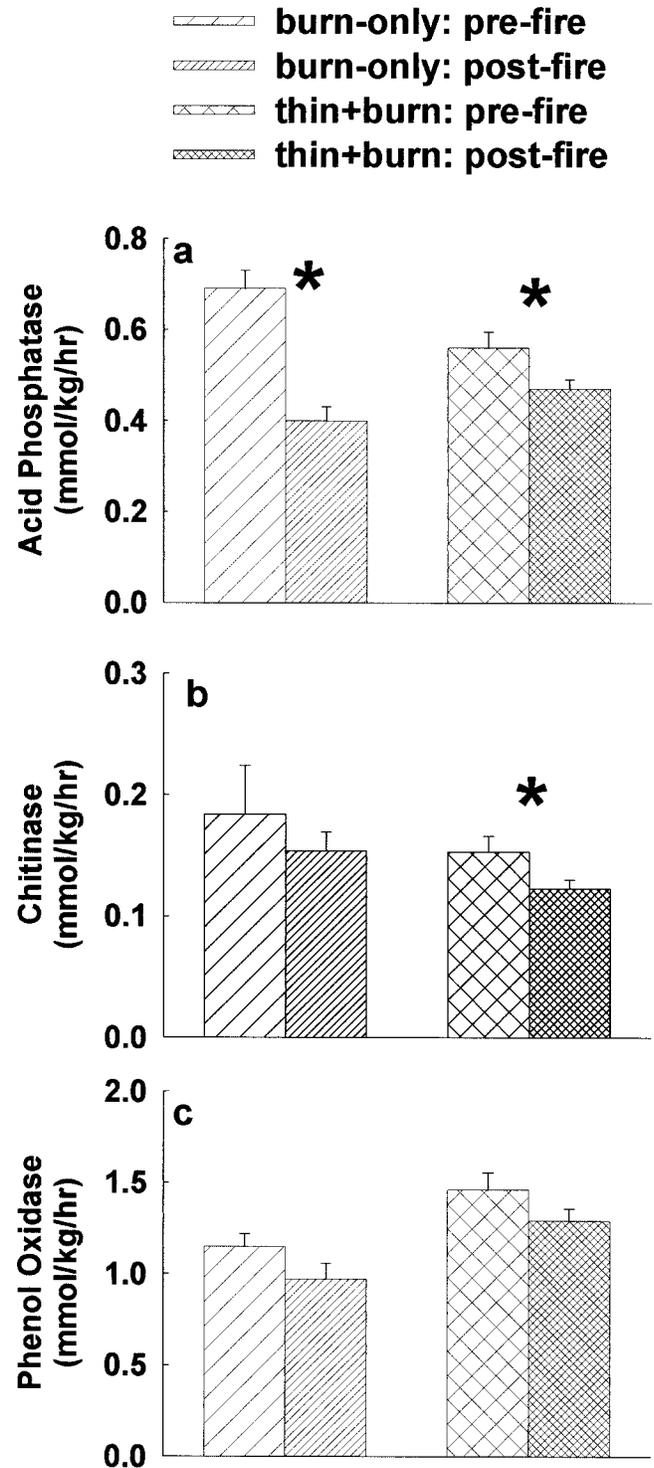


Fig. 3. Activity of acid phosphatase (a), chitinase (b), and phenol oxidase (c) before and 1 y after fire (2001 and 2002 for thin + burn treatment; 2002 and 2003 for burn-only treatment) at the Goosenest Adaptive Management Area, northern California. Histogram bars denote means with standard errors of the means; significant differences at $P = 0.05$ are indicated by asterisks.

ison with the existing fire-effects literature, most of which focuses on first-year effects.

When compared with pre-burn conditions, fire reduced soil organic matter quantity in the burn-only treatment, but not in the treatment units that had been mechanically thinned prior to burning. High pre-burn soil C content coupled with low fire severity may explain the lack of significant change in organic matter quantity in units that received thinning prior to prescribed fire. Other studies of mechanically thinned coniferous forests have also noted high soil C after logging, although post-harvest increases in soil C did not occur in the majority of studies reviewed by Johnson (1992). However, in two examples from mixed-conifer forests, increases in mineral soil C of 18% and 23% after whole-tree harvesting and 6 y after a clear-cut harvest, respectively, were reported; these increases may be attributed to the addition of slash to the forest floor, mixing of organic matter into mineral soil layers, and increased decomposition rates following harvesting (Johnson 1992).

The loss of soil organic matter is largely dependent upon the temperature and intensity of a fire (Ahlgren and Ahlgren 1960); prescribed burning generally results in increased soil C near the soil surface, but higher-severity fires have been shown to decrease soil C (Johnson 1992). Carbon can be added to the soil when charcoal from burned forest floor material is incorporated into the soil (Johnson 1992), and this can result in overly conservative estimates of the loss of soil organic C during a fire (Johnson and Curtis 2000). The significant loss of organic matter in the burn-only treatment could have been due either to loss via combustion during the fire or to increased microbial mineralization of organic matter after the fire (Johnson and Curtis 2000). As the results from our acid phosphatase activity assays do not support the latter, the more likely explanation appears to be greater consumption of organic matter by fire in the burn-only treatment than in the thin + burn treatment. Although fire weather and flame lengths were similar during the fires in the burn-only and thin + burn treatments, flame length (fire intensity) is not directly correlated with fire severity (loss of forest floor organic matter). For example, a slow-moving fire with low flame lengths may sustain greater soil heating and organic matter combustion than a relatively faster-moving fire with greater flame lengths (Busse et al. 2004).

More important to the difference in organic matter loss may have been the spatial continuity of the fuel bed. Our visual observations indicate that the activities associated with the thinning resulted in some mixing of litter, duff, and mineral soil in areas of particularly heavy vehicular and human traffic. This, in turn, could have resulted in a reduction of the exposure of some of that organic matter to direct combustion, thus preventing the prescribed fire from having as great an effect in the thin + burn treatment as it did in the burn-only treatment. Furthermore, although total pre-burn fine and coarse woody debris loads in the thin + burn treatment were equal to those of the burn-only treatment (both averaged 7.4 Mg ha^{-1}), fuels were not as

continuous in the thin + burn as in the burn-only treatment (Schmidt 2005).

We observed an increase in C:N ratio as a result of fire in the burn-only treatment but no corresponding change due to fire in the thin + burn treatment. In the burn-only treatment, the C:N ratio of the soil organic matter was likely decreased by the combination of an addition of partially combusted woody material and charcoal, both of which have high C:N ratio, and a loss of relatively low C:N ratio organic matter to direct combustion. The lack of a similar effect on C:N ratio in the thin + burn treatment could have resulted from lower fire severity (as discussed previously) or a greater proportion of relatively low C:N ratio organic matter in the fuels (the result of green needles being deposited on the surface as a result of thinning operations), or both.

The N mineralization rate was essentially unaffected by prescribed fire, whereas nitrification rate was reduced significantly in the thin + burn treatment but not in the burn-only treatment. The processes of N mineralization are accomplished by a wide variety of soil organisms, including bacteria, fungi, and nematodes, whereas nitrification is accomplished primarily by a specialist guild of bacteria. Thus, modification of the chemical and biochemical conditions in the soil is more likely to have affected the latter more than the former (Raison 1979).

It was initially unclear to us why there was a lack of change in N mineralization in the burn-only units, as many studies have demonstrated increases in N mineralization after single fires (reviews by Raison 1979, Boerner 1982, Wan et al. 2001). Such increases are often attributed to the alteration of organic matter by fire in such a manner as to render it more susceptible to microbial attack, to increases in microbial activity, and to changes in microclimate. To the degree to which our enzyme activity assays reflect components of microbial activity, our results are not consistent with an increase in microbial activity. Similarly, if one assumes that C:N ratio and other properties associated with organic matter quality (e.g., phenolic content, sclerophyll index) vary in parallel, the increase we observed in C:N ratio would suggest that susceptibility to microbial attack would have been reduced rather than enhanced by fire in this site. Given the paucity of growing-season precipitation in this region (monthly averages for the June–August sampling period were 0.76 cm/mo for 2002 and 0.25 cm/mo for 2003 [USFS 2006]), changes in microclimate during the parts of the year that are otherwise suitable for mineralization may not have been sufficient to overcome the lack of rainfall. Thus, the lack of stimulation of N mineralization by fire in this study may have reflected a lack of the types of changes in microbial activity, organic matter quality, and microclimate that have occurred in sites where increases in N mineralization have been demonstrated.

Total inorganic N in the soil solution increased as a result of fire in both treatments, although thin + burn plots exhibited a 4-fold increase in TIN from pre-fire to post-fire years while levels in burn-only plots mere-

ly doubled. The difference in the magnitude of the increase in TIN is consistent with studies that report that thinning and the creation of canopy gaps as small as 0.07 ha in forests create areas of greater N availability (Prescott et al. 1992, Parsons et al. 1994, Bauhus and Barthel 1995).

This increase in mineral N was expected, as Wan et al. (2001) concluded from their meta-analysis of fire effects on N that TIN generally increases in coniferous forests during the initial post-fire growing season. The difference in magnitude of increase observed for TIN may be explained by difference in fire severity as lower-intensity fires transfer larger amounts of NH_4^+ to the soil than more severe fires, during which a greater amount of soil N is volatilized (DeBano 1991). Mineral soil N may increase after fire due to transport of N from the forest floor (Wells et al. 1979), as well as from the release of NH_4^+ from soil minerals and clay-organic complexes during combustion, and both may be followed by conversion of NH_4^+ to NO_3^- due to nitrification (Russell et al. 1974, Raison 1979). However, such increases in available N are typically transitory and are likely to dissipate by the end of the second post-fire growing season (Wan et al. 2001).

We chose acid phosphatase as an indicator of overall microbial activity as the activity of this enzyme is often strongly correlated with microbial biomass (Kandeler and Eder 1993), microbial biomass N (Clarholm 1993), fungal hyphal length (Häussling and Marschner 1989), and N mineralization (Decker et al. 1999). Acid phosphatase activity was reduced significantly by both treatments. Similarly, Saa et al. (1993) reported 80–90% decreases in acid phosphatase levels after fire in gorse (*Ulex europaeus*) shrublands and pine plantations in Spain, and Boerner et al. (2000) reported decreases of similar magnitude to those we report here in their study of fire in mixed-oak (*Quercus* spp.) forests in eastern North America. Acid phosphatase activity may remain low for as long as 4 y after burning, at least in the jack pine (*Pinus banksiana*) forests of Ontario studied by Staddon et al. (1998).

In our study, acid phosphatase activity was reduced considerably more by the burn-only treatment (42%) than by the thin + burn treatment (16%). In a long-term (40+ y) study of microbial activity and soil organic matter in a Missouri oak flatwoods, Eivazi and Bayan (1996) demonstrated that more frequent burning causes greater reductions in acid phosphatase activity, β -glucosidase activity, and microbial biomass than did less frequent burning, and Staddon et al. (1998) demonstrated a direct correlation between fire severity and reduction in acid phosphatase activity in a jack pine forest. Thus, the greater reduction in acid phosphatase activity in our burn-only than the thin + burn treatment may again reflect greater effective fire severity in the former than the latter.

Chitinase activity reflects the use of chitin (the detrital remains of arthropods and fungi) as a source of both C and N by bacteria and, in some ecosystems, actinomycetes. As chitin is intermediate in its resistance to microbial metabolism, its synthesis is only induced when other, more labile C and N sources are

absent (Handzlikova and Jandera 1993). As chitinase is produced only by bacteria, changes in chitinase activity relative to that of other enzymes give an indication both of changes in the relative contribution of bacteria to microbial activity as well as changes in organic matter along the gradient from labile to recalcitrant (Handzlikova and Jandera 1993).

In our study, chitinase activity decreased as a result of the thin + burn treatment, but not as a result of fire alone. Such a decrease in chitinase activity should indicate a reduction in the importance of chitin as a source of C and N for bacterial production. As our results indicate that organic matter content and quality both decreased following the burn-only treatment but not the thin + burn treatment, one would anticipate that chitinase activity would increase in the burn-only treatment and remain unchanged in the thin + burn treatment. This is the opposite of what we actually observed. We currently have no mechanistic explanation for this apparent contradiction.

It should be noted, however, that the degree of variability present in chitinase activity in our pre-burn data for the burn-only treatment was 3- to 4-fold greater than was the case in any other enzyme-treatment-year combination. This variability was the reason that the difference between pre-burn and post-burn years was not statistically significant in the burn-only treatment, despite similar absolute ($0.03 \text{ mmol g}^{-1} \text{ h}^{-1}$ for both) and relative (16% and 20%) decreases in chitinase activity in the burn-only and thin + burn treatments. Whether this greater variability was a reflection of a difference in spatial heterogeneity between treatment units or an aberration in sampling and/or analysis is unclear.

The index of fungal activity we used was phenol oxidase, an enzyme produced primarily by white rot fungi, which is specific for highly recalcitrant organic matter such as lignin (Carlile and Watkinson 1994). Although phenol oxidase activity should not be considered a proxy for the abundance or activity of all fungi, it is a useful indicator of the activity of fungi that specialize on the breakdown of wood, bark, and other lignin-rich substrates (Carlile and Watkinson 1994). Based on this parameter, fungal activity was not affected by burning, alone or in combination with mechanical thinning. Given the preponderance of relatively recalcitrant materials in the forest floor and soils of this (and most other) coniferous forests and the relatively low severity of the fires, the lack of response by organisms that specialize on low-quality organic matter is not surprising. Taken together, our enzyme activity results suggest that the impact of fire may differ among types of soil organisms associated with decomposition, and those effects may be influenced by treatment type.

MANAGEMENT IMPLICATIONS AND CONCLUSIONS

This study demonstrates that fire has short-term effects on soil ecological properties, such as soil or-

ganic matter, N availability, and microbial enzyme activity, and that the combination of mechanical thinning and fire may have different effects from fire alone. Fire alone resulted in reduced soil organic matter content and increased C:N ratio, whereas the combination of a mechanical thinning treatment and burning did not affect soil organic matter. Activities associated with thinning may mix litter, duff, and mineral soil materials in such a manner as to make some of the organic matter less susceptible to combustion. Thus, the more continuous ground fuels and lower relative humidity in the burn-only treatment and the less continuous fuel bed in the more open thinned stand may have contributed to greater fire severity in the burn-only treatment than in the thin + burn treatment.

Soil organic matter is important to a productive forest ecosystem because of its role in stabilizing soil, maintaining soil conditions suitable for seedling establishment and growth, and supplying both nutrient storage and water-holding capacity. Thus, managers should be aware not only of the effect that management activities may have on soil organic matter, but also of the strong interactions that exist among stand density, fuel loading, fire behavior, and post-fire soil organic matter.

Our results also show that burning influences nitrification and available N in the soil, and that those effects differed between burn-only and thin + burn treatments. Based on prior studies in coniferous forests, the increases we observed in available N are likely to be transitory, and further study is required to determine if there will be longer-term effects of these treatments on the supply of N to trees.

The results of our assays of the activity of three enzymes that microbes secrete into the soil to digest soil organic matter are consistent with what one would expect if overall microbial activity was somewhat reduced by fire in these sites. The significant reduction in acid phosphatase and chitinase activities we observed suggest that there may have been a decrease in the activity of microbes that rely on more easily digested (labile) organic matter, and the lack of change in phenol oxidase activity suggests that the activity of organisms that specialize on recalcitrant organic compounds (such as wood-rotting fungi) did not likely change to any appreciable degree. We assume that whatever effect these fires had on microbial community structure is likely to dissipate as new litterfall restores the pre-fire nutritional characteristics of the forest floor and soil organic matter, though longer-term studies are required to verify this.

Prescribed fire following thinning resulted in a greater impact on nitrification and soil enzyme activities than did prescribed fire alone, whereas changes in soil organic matter content and C:N ratio were greater as a result of fire alone. Thus, for our mixed-conifer site in northern California, the application of prescribed fire to mechanically thinned stands has a greater impact on soil biological processes 1 y after burning, whereas soil C and organic matter quality are impacted more by fire in stands that were not mechanically thinned prior to burning. This study pro-

vides information useful for predicting the short-term, ecosystem-level effects of different forest restoration and wildfire hazard-reduction techniques in the widespread ponderosa pine ecosystem type.

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RESPONSE OF FOUR LAKE WALES RIDGE SANDHILL SPECIES TO REINTRODUCTION OF PRESCRIBED FIRE WITH AND WITHOUT MECHANICAL PRE-TREATMENT

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ABSTRACT

Restoration of endemic-rich but long-unburned sandhills on the Lake Wales Ridge (LWR) of peninsular Florida poses a considerable challenge to land managers and plant conservationists. Decades of fire suppression have led to the development of an extensive oak-dominated subcanopy and a concomitant loss of graminoid fine fuels and herb diversity. Two major objectives of LWR sandhill restoration are 1) protection of the sparse longleaf pine (*Pinus palustris*) canopy, which provides an irreplaceable source of fine fuels through annual needle-drop; and 2) promotion of the many plant species ($n > 15$) endemic to this globally imperiled ecosystem. As part of a larger experiment to investigate the recovery dynamics of a long-unburned LWR sandhill following the reintroduction of fire with and without prior chainsaw-felling of the subcanopy (saw + burn versus burn-only treatments), we conducted pre- and post-treatment censuses of 228 longleaf pines and of hundreds of individuals of three species of federally listed plants: scrub plum (*Prunus geniculata*), a multistemmed shrub; and two herbs, scrub buckwheat (*Eriogonum longifolium* var. *gnaphalifolium*) and Lewton's milkwort (*Polygala lewtonii*). We found that 1) longleaf pine mortality was significantly higher in the saw + burn treatment, most likely due to higher fire temperatures relative to the burn-only treatment; 2) scrub plum post-burn resprouting rates were independent of treatment (and largely independent of fire severity); 3) scrub buckwheat appears to have benefited most from the higher fire intensity and more complete fire coverage in the saw + burn treatment, demonstrating higher rates of seedling recruitment and flowering than in the burn-only treatment; and 4) Lewton's milkwort seedling recruitment and survival were significantly greater in burned versus unburned quadrats. These results suggest that while some sandhill endemics benefit from the mechanical pre-treatments used to facilitate the reintroduction of fire, care must be taken to prevent loss of longleaf pines, particularly in pre-treated areas with their greater fire intensities.

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EFFECTS OF MECHANICAL TREATMENTS AND FIRE ON LITTER REDUCTION IN FLORIDA SCRUB AND SANDHILL

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ABSTRACT

Florida scrub, a xeric shrubland, and Florida sandhill, a xeric savanna, are fire-maintained ecosystems threatened by habitat loss and fire suppression. Mechanical treatments such as logging, mowing, and chainsaw felling of the subcanopy (vegetation 3–8 m high) are being used as alternatives to prescribed burning or in combination with burning to restore Florida ecosystems. We focused on litter cover as a key response variable because continuous litter cover in unburned areas of Florida scrub and sandhill adversely affects populations of rare endemic herbs by reducing seedling recruitment and may also reduce overall herb diversity. We initiated three experiments to evaluate the effects of mechanical treatments, with and without fire, on litter cover: “log and burn” and “mow and burn” in scrub; and “saw and burn” in sandhill.

Logging or mowing alone or combined with fire, and subcanopy felling combined with fire, had different effects on litter cover. Logging caused significant reductions in litter cover 2 y after treatment, while burning without prior logging caused only modest (nonsignificant) decreases. However, in contrast to the burn-only treatment, logging also resulted in large areas of soil disturbance, particularly along skid trails used for removing logs; these areas were rapidly colonized by weedy native species. Although mowing alone did not reduce litter cover, mowing was effective as a pre-treatment to burning, provided that the burn took place shortly after mowing. The mow-and-burn treatment reduced litter cover in one site when fire followed mowing by 1 mo, but not in another site where fire was delayed by 1 y. Chainsaw felling of the oak-dominated subcanopy followed by burning reduced litter cover significantly, while burning alone failed to do so. Subcanopy felling may be an advantageous pre-treatment for restoration of long-unburned sandhill.

We do not recommend mechanical treatments without fire for restoration of fire-suppressed Florida scrub or sandhill. Mechanical treatments followed by fire can be effective in removing litter and thereby creating bare sand microsites that are key to rare species recruitment and herb diversity. However, mechanical treatments should be applied cautiously and results should be closely monitored. Prescribed fire remains the preferred method for restoring and maintaining Florida scrub and sandhill and their rare plant species.

keywords: Florida sandhill, Florida scrub, logging, mowing, prescribed burning, restoration.

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INTRODUCTION

Florida scrub, a xeric shrubland, and Florida sandhill, a xeric savanna, are fire-maintained ecosystems threatened by habitat loss, habitat fragmentation, and fire suppression (Menges 1999). Florida scrub and sandhill are rich in endangered and endemic plants (Christman and Judd 1990, USFWS 1999, Coile and Garland 2003), many of which are prominent in gaps among dominant shrubs and grasses. Fire removes litter and top-kills woody plants, opening up bare sand patches that provide critical habitat for many rare and imperiled species (Hawkes and Menges 1996, Menges and Kimmich 1996, Quintana-Ascencio et al. 2003). Long-unburned sites have declining populations of these species (Menges and Kohfeldt 1995), along with few bare sand patches and abundant subcanopy (3–8 m tall) oaks (*Quercus* spp.) and canopy pines (*Pinus* spp.). Historically, lightning-ignited fires in sandhills were frequent (2–10 y) and of low intensity, whereas lightning-ignited Florida scrub fires were infrequent (10–80 y) and of higher intensity (Abrahamson et al. 1984, Myers 1985).

Much of the remaining long-unburned scrub and

sandhill habitats occur near developed areas, where land managers may be hesitant to use prescribed fire because of potential control problems or smoke management concerns. Mechanical treatments such as logging, mowing, and roller chopping are being used across Florida as surrogates for burning or as pre-treatments to burning (Greenberg et al. 1995, Roberts and Cox 1999, Berish and Kubilis 2002, Schmalzer et al. 2003). Burns in mechanically treated areas may be easier to prescribe and conduct.

Studies of mechanical treatments in Florida scrub have mostly focused on the response of shrub cover and/or shrub height or have not always included a burn-only treatment (Roberts and Cox 1999, Berish and Kubilis 2002, Schmalzer et al. 2003). One such study (Berish and Kubilis 2002), conducted at the Lake Wales Ridge Wildlife and Environmental Area, concluded that mowing or mowing plus burning caused no negative effects in scrub; however, this study did not include a burn-only treatment or pre-treatment data. Another study of logging at Ocala National Forest also did not include a burn-only treatment (Greenberg et al. 1995). Roberts and Cox (1999) compared burning with mechanical treatments and burning in scrub and found that, with or without a mechanical pre-treatment, fire was essential for scrub manage-

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ment. Schmalzer et al. (2003) found that mechanical treatments were as effective as fire in reducing shrub height. In sandhill ecosystems, frequent fire is necessary to support high species diversity (e.g., Glitzenstein et al. 2003, Provencher et al. 2003), but introduction of fire to long-unburned sandhills can have negative consequences such as excessive pine mortality (Varner et al. 2005). Consequently, mechanical removal of hardwood subcanopies has been used as an alternative to fire. However, in the most thorough study, with many treatments and a comparison to reference sites, fire alone was the least expensive and most effective treatment for increasing species richness and herbaceous cover (Provencher et al. 2001). Although fire is also important to restoring fire-suppressed sandhills in peninsular Florida (Reinhart and Menges 2004), no work has been published on the effects of mechanical treatments on sandhill restoration in this region.

Despite the increasing application of mechanical treatments alone or combined with fire, we still do not have a good answer to key questions: Can mechanical treatments achieve the ecological benefits of fire, or will they cause unintended harmful effects to vegetation and rare species? Can mechanical treatments and fire be used together to speed up restoration of long-unburned areas? Our goals in restoring fire-suppressed Florida scrub and sandhill, regardless of whether fire, mechanical treatments, or their combination are used for restoration, are to reduce shrub height and cover and to remove litter to provide bare sand habitat for the rare herbaceous species endemic to these ecosystems (Christman and Judd 1990, Menges 1999).

To address these questions, we conducted three experiments: "log and burn," "mow and burn," and "saw and burn." The log-and-burn experiment studied the effects of logging and its compatibility with managing Florida scrub vegetation and its endemic plants. The mow-and-burn experiment examined the effects of mowing used as a fire surrogate and as a pre-treatment to fire on the regeneration of Florida scrub at two sites. The saw-and-burn experiment appraised the effects of chainsaw felling of the subcanopy followed by burning and burning alone as restoration treatments for Florida sandhill. In this paper, we focus on changes in litter cover because the removal of litter by fire is key to the recruitment and persistence of many listed plant species (Hawkes and Menges 1996, Menges and Kimmich 1996, Quintana-Ascencio et al. 2003) and because mechanical treatments have the potential to accumulate rather than to remove litter.

STUDY AREA

Our studies of mechanical treatments in Florida scrub and sandhill are from long-term experiments taking place on four long-unburned sites on the Lake Wales Ridge of central peninsular Florida. The log-and-burn experiment took place on the Lake Wales Ridge State Forest, Arbuckle tract (27.67°N, 81.42°W). The Arbuckle study site includes both rosemary phase

and oak phase sand pine scrub (sensu Abrahamson et al. 1984); in rosemary scrub the dominant shrub is Florida rosemary (*Ceratiola ericoides*), while in oak scrub the dominants are several species of clonal shrubby oaks. The mow-and-burn experiment occurred in the Lake Wales Ridge Wildlife and Environmental Area, Lake Apthorpe Scrub (LAS) (27.36°N, 81.33°W) and Lake Placid Scrub (LPS) (27.22°N, 81.394026°W). The LAS and LPS study sites are rosemary phase Florida scrub. The saw-and-burn experiment took place on the Lake Wales Ridge National Wildlife Refuge, Carter Creek tract (27.54°N, 81.41°W). The Carter Creek study site is Florida sandhill on xeric yellow sand with sparse longleaf pine (*Pinus palustris*) cover.

METHODS

Treatments

We compared three commonly used land management treatments for restoring xeric upland Florida ecosystems. The log-and-burn experiment included log-only, burn-only, log-and-burn, and control treatments. Between December 1998 and January 1999, all mature sand pines (*Pinus clausa*) in the logging treatment areas were removed by a commercial logging company. Following the standard practice, trees were "de-limbed" on-site and dragged by heavy equipment to loading areas. Remaining slash was left in discrete piles, and the removal of logs resulted in considerable soil disturbance associated with skid trails. Prescribed burning in the log-and-burn treatment plots in August 1999 was interrupted by deteriorating fire weather. A second prescribed burn in September 2000 included the burn-only plots and the remaining log-and-burn plots. In both burns, firing techniques included head-and-strip headfires after the initial establishment of a blackline on the downwind perimeter of the area being burned.

The mow-and-burn experiment included mow-only, burn-only, mow-and-burn, and control treatments. Mowing was completed in June 1999 (LAS) and in July 2000 (LPS) with a Brown tree cutter (Brown Manufacturing Corp., Ozark, AL). The burn at LAS took place 1 mo after mowing (June 1999) and burned quickly and completely. The burn at LPS was conducted 1 y after mowing (July 2001) due to logistical constraints; this site burned slowly and patchily. Firing techniques were similar to the log-and-burn experiment.

The saw-and-burn experiment had burn-only, saw-and-burn, and control treatments; a saw-only treatment was not included because this treatment is not typically used by Florida land managers. We used chainsaws to fell oaks and other subcanopy (3–8 m tall) trees (except for longleaf pines) in a 15-m "fuel-enhancement zone" centered on each of 24 community plots within the saw-and-burn treatment area. This study site was prescribed burned in July 2001, 1 mo after chainsaw felling (May–June 2001). Backing and flanking fires

were ignited, followed by strip headfires throughout the burn units.

Date Collection and Analysis

Although sampling protocols varied among projects due to the nature of the vegetation and treatments, in all experiments we estimated litter cover in small plots or using line intercepts. In the log-and-burn experiment, we sampled litter cover within 24 50 × 25-m plots pre-treatment and 1, 2, and 5 y post-treatment. In the mow-and-burn experiment, we sampled 80 20 × 2-m belt transects, measuring litter cover along each transect pre-treatment and 1, 2, and 5 y post-treatment. In the saw-and-burn experiment, we sampled 61 5-m-radius community plots; 11 plots in the burn-only treatment did not burn and were excluded from the study. Within each community plot, we estimated litter cover in eight 0.19-m² quadrats. We sampled pre-treatment and 1, 2, 3, and 4 y post-treatment.

We summed litter cover data collected in herb quadrats (log-and-burn, saw-and-burn) or along line intercepts (mow-and-burn) to the community plot level. We used pairwise Kruskal–Wallis tests to compare the change in percent litter cover pre-treatment versus 2 y post-treatment (mow-and-burn, log-and-burn) or 1 y post-treatment (saw-and-burn) because the data were not normally distributed. Degrees of freedom = 1 for all statistics and χ^2 values are Kruskal–Wallis chi-square values.

Species names follow Wunderlin and Hansen (2003).

RESULTS

Log-and-Burn Experiment

Logging treatments were effective in reducing litter cover (Figure 1). Litter cover was reduced more in the log-only and the log-and-burn treatments reduced litter cover significantly lower than the control ($\chi^2 = 9.00$, $P = 0.003$; $\chi^2 = 7.50$, $P = 0.006$, respectively). Litter cover reductions were only somewhat lower but not different in the burn-only versus the control treatment ($\chi^2 = 0.410$, $P = 0.522$); the log-and-burn versus the log-only treatments were not different ($\chi^2 = 0.798$, $P = 0.372$).

Mow-and-Burn Experiment

At LAS, where burning was done 1 mo after mowing, litter cover was reduced in both the mow-and-burn and burn-only treatments compared with the control ($\chi^2 = 27.774$, $P < 0.001$; $\chi^2 = 13.605$, $P < 0.001$, respectively), while the litter cover in the mow-only treatment was not different from the control ($\chi^2 = 1.467$, $P = 0.226$; Figure 2A). However, at LPS, where burning was delayed for a year after mowing, the mow-and-burn treatment was ineffective in reducing litter cover ($\chi^2 = 2.599$, $P = 0.107$; Figure 2B). The burn-only treatment effectively reduced litter cover versus all other treatments at LPS ($P \leq 0.001$; Figure 2B).

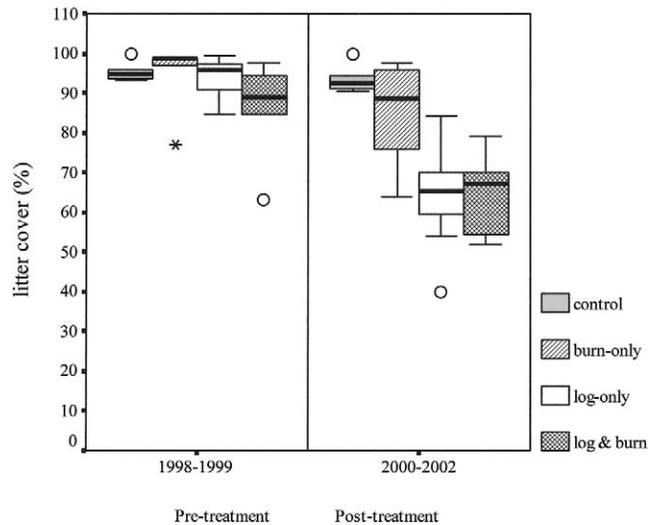


Fig. 1. Boxplots showing mean percent litter cover pre-treatment (1998–1999) and 2 y post-treatment (2000–2002) for a log-and-burn experiment to restore fire-suppressed scrub, Lake Wales Ridge, central Florida. The lower and upper horizontal bars of the boxplot represent the 25th and 75th percentiles, respectively; the solid middle bar represents the median. The lower and upper whiskers show the largest and smallest values that are not outliers. The circles are outliers (more than 1.5 box-lengths from 25th and 75th percentiles) and the asterisks are extreme outliers (more than 3 box-lengths from 25th and 75th percentiles).

Saw-and-Burn Experiment

The saw-and-burn treatment was most effective in reducing litter cover (Figure 3). Litter cover in the saw-and-burn treatment decreased significantly compared with both the burn-only and the control treatments ($\chi^2 = 15.478$, $P < 0.001$; $\chi^2 = 26.906$, $P < 0.001$, respectively). The burn-only treatment marginally reduced litter cover ($\chi^2 = 3.762$, $P = 0.052$).

DISCUSSION

By the 1980s, >85% of fire-maintained Florida scrub and sandhill ecosystems on the Lake Wales Ridge of central peninsular Florida had been lost to development (Peroni and Abrahamson 1985). Much of the remaining Florida scrub and sandhill (currently <15% of its original area) is embedded in an increasingly urbanized landscape, making it difficult for land managers to apply prescribed fire safely. Remaining xeric uplands in other parts of the state are even rarer and more fragmented. Thus, land managers have turned to various mechanical means as either surrogates for fire or as pre-treatments to fire. When used as pre-treatments to fire, mechanical treatments reduce the height of fuel and make it more homogeneous, decreasing the risk of fire escape and facilitating even burn coverage. Based on preliminary results from three studies on four sites, we have found that some mechanical treatments are more effective than others in creating conditions that promote the restoration and maintenance of Florida scrub and sandhill ecosystems.

Logging reduced litter cover much more than

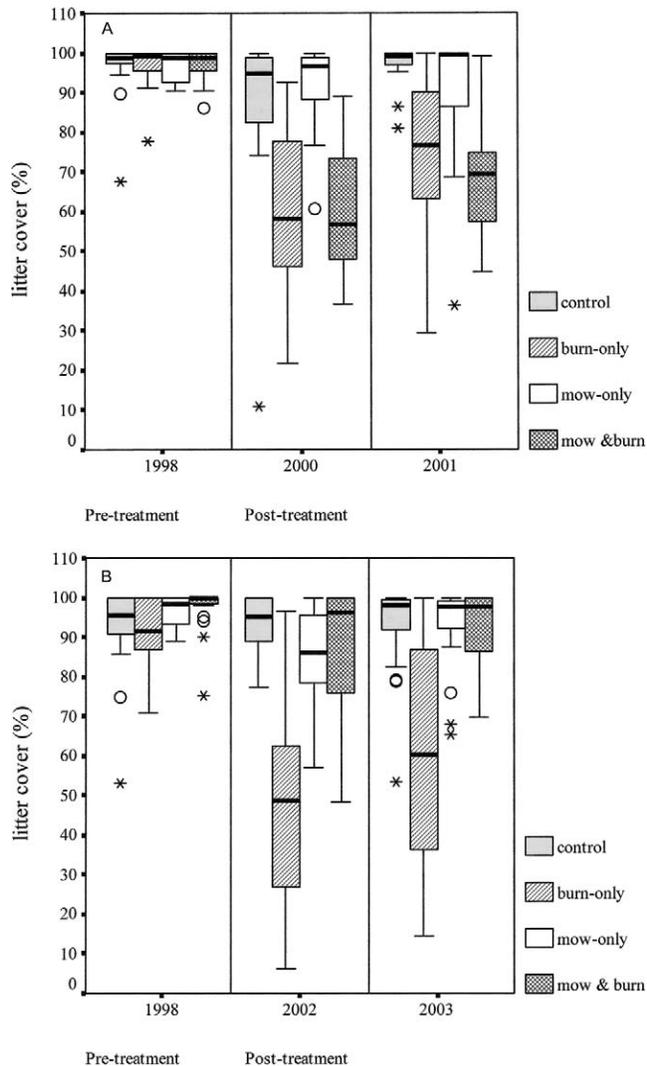


Fig. 2. Boxplots comparing percent litter cover pre-treatment (1998) and post-treatment (2000–2003) for a mow-and-burn experiment to restore fire-suppressed scrub, Lake Wales Ridge, central Florida. (A) Lake Apthorpe scrub. (B) Lake Placid scrub. The lower and upper horizontal bars of the boxplot represent the 25th and 75th percentiles, respectively; the solid middle bar represents the median. The lower and upper whiskers show the largest and smallest values that are not outliers. The circles are outliers (more than 1.5 box-lengths from 25th and 75th percentiles) and the asterisks are extreme outliers (more than 3 box-lengths from 25th and 75th percentiles).

burning alone; however, skid trails from timber removal in the log-only and log-and-burn treatments created large areas of bare sand characterized by considerable soil disturbance. Logged areas were dominated by weedy natives and are susceptible to invasion by exotics such as cogongrass (*Imperata cylindrica*) and natal grass (*Rhynchelytrum repens*). Logging produced more extreme changes than burning alone in the canopy layer (M.A. Rickey, E.S. Menges, and C.W. Weekley, unpublished data) and in litter cover. All treatments were successful in reducing the subcanopy layer and reducing shrub cover (M.A. Rickey, E.S. Menges, and C.W. Weekley, unpublished data). Because of the soil disturbances associated with logging, we recom-

mend caution when logging in Florida scrub. Over time, the negative effects of these soil disturbances may decline (P.A. Schmalzer, Dynamac Corporation, personal communication).

The effectiveness of the mow-and-burn treatment depended upon the timing of the burn. When the burn occurred 1 mo after mowing, the mow-and-burn treatment mimicked the burn-only treatment; when the burn was delayed for a year, litter reductions in the mow-and-burn treatment were more similar to mowing alone, which was not effective in reducing litter cover. This result is in agreement with that of Schmalzer et al. (2003), who found that mechanical treatments plus burning were most similar to burning alone when the fire occurred <6 mo after cutting. In our study, the mow-only treatment was not effective because it created a thick layer of thatch that reduced bare sand patches needed for seedling requirement by many rare Florida scrub plants. All treatments at both mow-and-burn sites were effective in reducing shrub heights relative to the control (M.A. Rickey, E.S. Menges, and C.W. Weekley, unpublished data). Chainsaw felling followed by fire was more effective in reducing litter cover than fire alone, perhaps because of the higher fire intensity (Wally et al. 2006) and more complete coverage (E.S. Menges, personal observation) associated with the chainsaw pre-treatment. All three mechanical treatments examined in this study, when used as pre-treatments for prescribed burning, were at least as effective in reducing litter cover as burning alone.

MANAGEMENT IMPLICATIONS

Mechanical treatment followed quickly by fire can be a useful management tool to accelerate restoration of Florida scrub and sandhill sites degraded by decades of fire suppression. Because of the scarcity of data on the effects of various mechanical treatments or pre-treatments, we strongly recommend pre- and post-treatment monitoring of vegetation responses to the methods employed. This is particularly important if mechanical treatments are used as a surrogate for fire. Monitoring programs should explicitly target the rare herbaceous species endemic to Florida scrub and sandhill because they may be vulnerable to mechanical treatments. However, because these endemics are also especially vulnerable to fire suppression, the use of mechanical treatments followed by fire may be of benefit. Soil disturbance may lead to increases in exotic plants, or as in the case of our log-and-burn experiment, weedy native species. Mechanical treatments combined with fire may be especially useful for the first attempt to burn long-unburned sites; however, subsequent management of Florida scrub and sandhill should focus on prescribed burning.

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We thank Archbold Biological Station, the Florida Division of Forestry, the Florida Fish and Wildlife Conservation Commission, The Nature Conservancy,

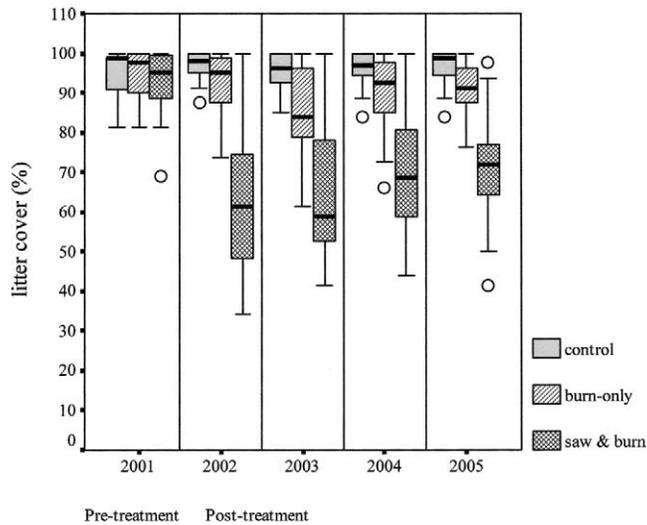


Fig. 3. Boxplots comparing mean percent litter cover pre-treatment (2001) and post-treatment (2002–2005) for a saw-and-burn experiment to restore fire-suppressed sandhill habitats, Lake Wales Ridge, central Florida. The lower and upper horizontal bars of the boxplot represent the 25th and 75th percentiles, respectively; the solid middle bar represents the median. The lower and upper whiskers show the largest and smallest values that are not outliers. The circles are outliers (more than 1.5 box-lengths from 25th and 75th percentiles) and the asterisks are extreme outliers (more than 3 box-lengths from 25th and 75th percentiles).

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A COMPARISON OF NATIVE VERSUS OLD-FIELD VEGETATION IN UPLAND PINELANDS MANAGED WITH FREQUENT FIRE, SOUTH GEORGIA, USA

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ABSTRACT

Fire-maintained, herb-dominated upland pinelands of the southeastern U.S. Coastal Plain may be broadly divided into those that have arisen through secondary succession following abandonment of agriculture (old-field pinelands) and those that have never been plowed (native pinelands). The ability to distinguish these habitat types is important for setting conservation priorities by identifying natural areas for conservation and appropriate management and for assessing the ecological value and restoration potential for old-field pine forests managed with frequent fire. However, differences in species composition have rarely been quantified. The goals of this study were to characterize the species composition of native and old-field pineland ground cover, test the ability to distinguish communities of previously unknown disturbance history, and suggest indicator species for native versus old-field pinelands. Plant composition was surveyed in areas known to be native ground cover, those known to be old fields, and those with an uncertain disturbance history. Twelve permanent plots were established in each cover type and sampled in spring (April–May) and fall (October–November) in 2004 and 2005. Of the 232 species identified in the plots, 56 species were present only in native ground-cover plots, of which 17 species occurred in a sufficient number of plots to have a statistically significant binomial probability of occurring in native ground cover and might be considered indicator species. In addition, 15 species were confined to old fields, of which 5 had a statistically significant binomial probability. Additionally, plant census transects from a previous survey were comparatively analyzed, yielding a total of 432 species, of which 111 were present only in native ground-cover transects and 3 occurred in a sufficient number of transects to have statistically significant binomial association with native areas. Also, 111 species were confined to old fields but none in a sufficient number of transects for a significant association with old fields. In both the plot and transect data sets, most old-field species represented a subset of those found in native areas, suggesting differential ability of certain native species to disperse to and become established in abandoned agricultural land. These results will assist in identifying natural areas for conservation as well as assessing the ecological value and restoration potential of old-field pine forests managed with frequent fire.

keywords: indicator species, longleaf pine, native ground cover, old fields, *Pinus palustris*, prescribed fire, secondary succession, South Georgia, upland pinelands.

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INTRODUCTION

A detailed knowledge of a region's natural communities is crucial for conservation and restoration efforts as well for the management of currently protected areas. This baseline knowledge allows for a determination of the degree to which a degraded system has changed and potentially concurrent loss of biodiversity and possible changes in ecosystem processes (Noss 2000). The ability to identify undisturbed natural communities may be obfuscated by a past history of anthropogenic disturbance or alteration of natural disturbance regimes. In the southeastern U.S. Coastal Plain, intensive agriculture is chief among anthropogenic activities that have contributed to a 97% conversion of native pineland habitat (Frost 1993). Thus, conservation efforts require the means to distinguish and characterize native versus post-agricultural communities in order to set conservation priorities for acquisition and

protection and to assess the ecological value and restoration potential of each.

Following the large-scale abandonment of agricultural fields on the Coastal Plain in the late 19th and early 20th centuries (Paisley 1968, Brueckheimer 1979), fallow fields that were periodically burned returned to pinelands with an herb-dominated understory (Moser et al. 2002). Thus, fire-maintained, herb-dominated upland pinelands of the southeastern U.S. Coastal Plain may be broadly divided into those that have arisen through secondary succession following abandonment of agriculture (old-field pinelands) and those that have never been plowed (native pinelands). With proactive management (i.e., prescribed burning and selective timber thinning), these old-field pinelands provide habitat to gopher tortoises (*Gopherus polyphemus*), Bachman's sparrows (*Aimophila aestivalis*), fox squirrels (*Sciurus niger*), and reintroduced red-cockaded woodpeckers (*Picoides borealis*) (Masters et al. 2003). Even so, old-field successional habitats are thought to constitute a major shift in floristic characteristics from their original composition (Means and Grow 1985, Myers 1990). In the eastern portion of the southeastern U.S. Coastal Plain, upland old-field

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pinelands once dominated by longleaf pine (*Pinus palustris*) and wiregrass (*Aristida stricta*) are now typically dominated by loblolly pine (*Pinus taeda*) and shortleaf pine (*P. echinata*) (Moser et al. 2002) and have lost wiregrass as a component of their herbaceous ground cover (Crafton and Wells 1934, McQuilken 1940, Oosting 1942, Grelen 1962, Clewell 1986, Hedman et al. 2000). It is generally observed that old-field vegetation has a smaller number of native species and that these species are relatively common (Clewell 1986, Hedman et al. 2000, Kirkman et al. 2004). This prevalence is attributed to their ability to disperse to and colonize disturbed soil (Kirkman et al. 2004). Thus, old-field pinelands may be placed at a lower priority for conservation and protection.

Although species composition of native longleaf pine habitats has often been described (e.g., Bridges and Orzell 1989, Hardin and White 1989, Drew et al. 1998, Kush and Meldahl 2000, Varner et al. 2003, Carter et al. 2004), comparatively few published studies have compared vegetation composition between old-field and relatively undisturbed upland pine habitats (Hedman et al. 2000, Kirkman et al. 2004). These studies have shown significant differences in species composition between old-field sites and "reference sites" representing the native community type, as well as lower species richness in old-field sites because of dispersal limitations on certain native species (Kirkman et al. 2004). Native pineland communities are also sensitive to vehicular traffic, soil compaction, and surface scarification associated with planted pine site preparation, logging, and military training (Hedman et al. 2000, Dale et al. 2002, Smith et al. 2002). However, soil disturbance associated with intensive agriculture appears to have the greatest impact on species composition in fire-maintained pinelands relative to these other disturbance types (Hedman et al. 2000).

The sensitivity of many species to anthropogenic disturbance makes them useful indicators of habitat integrity (Noss 1990, Kimberling et al. 2001, McLachlan and Bazely 2001, Moffatt and McLachlan 2004). Indicator species should be sensitive to the environmental stress of interest, and thus may indicate the biological integrity of an ecosystem (Dale and Beyeler 2001) and abundant and tractable components of the system (Welsh and Ollivier 1998). Indicator species for relatively undisturbed native pinelands maintained with frequent fire in the Coastal Plain have been proposed in other studies (Rodgers and Provencher 1999, Dale et al. 2002, Smith et al. 2002, Kirkman et al. 2004). However, additional studies are needed both to confirm the reliability of certain widely distributed species as indicators throughout the region as well as to identify local indicator species among physiographic features within the region (e.g., Dougherty Plain, Tifton Uplands).

The goals of this study were to characterize and contrast the species composition between native and old-field upland clayhill pinelands in Southwest Georgia, to test the utility of these data in interpreting the community integrity of sites with unknown disturbance history, and to identify possible indicator species that can aid in distinguishing these community types. The results should

assist in identifying natural areas for conservation and appropriate management and for assessing the ecological value and restoration potential for old-field pine forests managed with frequent fire.

METHODS

Study Area

The study was conducted within the Red Hills region of southern Georgia and northern Florida on Pebble Hill Plantation (PHP) (30°35'N, 84°20'W), which covered approximately 1,222 ha in Grady and Thomas counties, Georgia. The Red Hills region was characterized by gently sloping, well-drained sandy or loamy soils underlain by clayey or sandy sub-horizons (Calhoun 1979). Plots were more or less evenly distributed among the following soil types: Bonneau loamy sand, Dothan loamy sand, Lucy loamy sand, Tifton loamy sand, Faceville sandy loam, and Nankin-Cowarts sandy loam. Mean annual temperature was 19.6°C (11°–27.4°C monthly means) and mean annual precipitation was 1,373.4 mm (Southeast Regional Climate Center 2004). The growing season for this region was from early March to November (Calhoun 1979; T.E. Ostertag and K.M. Robertson, unpublished data). PHP had been managed for northern bobwhite (*Colinus virginianus*) and timber during the past century, primarily with the use of frequent prescribed fire (1- to 2-y fire interval) and both even- and uneven-aged management systems.

The forested upland habitats at PHP were a mixture of old fields, pine plantations, and native pinelands (never plowed) (Robertson and Ostertag 2003). The native areas had a canopy dominated by longleaf pine, often mixed with shortleaf pine, and an understory supporting a high diversity of other woody plants, forbs, and grasses, especially wiregrass. The old-field pineland habitats had a canopy dominated by some mixture of shortleaf and loblolly pines and an understory of woody species typical of disturbed areas in the region, such as water oak (*Quercus nigra*) and sweetgum (*Liquidambar styraciflua*), and herbs composed of some subset of native species as well as agricultural weeds. Timber management had been similarly applied to the specifically selected native and old-field pineland sites, such that stand densities and structure were similar between the two habitat types (average of 9.6 ± 6.3 SD m²/ha in native stands based on 54 random plots, $10.9 \text{ m}^2 \pm 6.3 \text{ m}^2$ basal area/ha in old-field stands based on 59 random plots [K.M. Robertson, unpublished data]).

Site Selection and Sampling Methods

Plant communities were compared among the following cover types: native ground cover, old fields, and areas of unknown soil disturbance history. Native ground cover was identified by the presence of wiregrass, based on the general observation that wiregrass does not readily return to extensive areas of heavily disturbed soil or prolonged fire suppression (Hebb

1957, 1971; Woods 1959; Grelen 1962; Harris et al. 1974; Schultz and Wilhite 1974; White et al. 1975; Myers 1990). Old-field sites were identified as cultivated in a 1928 (earliest known) aerial photograph. Based on the age of current pines and other historical records, we estimated these sites had been abandoned from cultivation circa 1950. Areas of unknown management history (may or may not be native ground cover) were forest at the time of the 1928 aerial photograph (current age of trees approximately 150 y old) but lacked wiregrass at the time of this study.

For each of the three cover types, 12 permanent sampling plots were established, with 3 plots in each of 4 separate burn units. Burn units ranged in area from 2.6 to 18.5 ha. Burn units and plot locations within burn units were randomly chosen using ArcView 3.2 Animal Movement extension (Environmental Systems Research Institute 1999). Potential plot locations were limited to ridge tops and the upper halves of slopes to restrict them to upland pine habitats. Each plot was 100 m² (10 m × 10 m), with nested subplots of 10 m² and 1 m² in one corner. Thus, the 36 total plots covered 3,600 m². Species were censused starting with the 1-m² subplot, then proceeding to the 10-m² and the 100-m² plots. Species of all herbaceous vascular plants and woody shrubs and trees <1 m high were censused within the plots. Percent cover within the 100-m² plot was estimated for each species using a modified Daubenmire cover class method (Peet et al. 1998). Maximum cover for each species was 100%, but cumulative cover for all species could exceed 100%. The plots were censused four times between fall 2003 and spring 2005 (once in each of the following: October–November 2003, October–November 2004, April–May 2004, and April–May 2005) to incorporate seasonal and annual variation in the presence and visibility of plant species. Plants were identified to species and otherwise were not included in the analysis. Unidentified plants were generally seedlings or those badly damaged by herbivory or senescence. Unidentified specimens accounted for 8% of all samples collected.

Species composition in the study plots was compared to species lists compiled between July 1995 and November 2002 from transects running throughout PHP (A. Gholson and C. Martin, PHP, unpublished data). Transects used in the analysis were limited to those entirely contained within native ($n = 9$) and old-field ($n = 8$) cover types. There were no transects contained entirely within the unknown-disturbance-history cover type. These censuses incorporated plants that could be easily observed by the surveyors to either side of each transect (total width of 4 m). Transects varied in length from 160 to 719 m due to size of the burn unit. The total area censused for all transects was approximately 22,000 m². Transects were not bound by topographic or hydrological constraints as were the study plots, such that lower hillslope and wetland pine-lands were included. All plants, including trees and shrubs regardless of size, were recorded. Taxonomy and nomenclature for both the plot censuses and transects follows Wunderlin and Hansen (2003).

Data Analysis

A detrended canonical correspondence analysis (DCCA) in CANOCO 4.5 (ter Braak and Smilauer 2002) was used to test the null hypothesis of no predictable difference in species composition and cover among native, old-field, or unknown study plots. The median cover values for 100-m² plots estimated for species were used as the response variables in the analysis, and plots served as units of replication. Monte Carlo permutations (499 iterations) on the first canonical axis were used to produce an F -statistic testing for differences among sampling plots (Leps and Smilauer 2003). A detrended correspondence analysis (DCA) in CANOCO 4.5 was used to create an ordination diagram in CANODRAW 4.5 (ter Braak and Smilauer 2002) to visually assess floristic similarity among sampling plots (Leps and Smilauer 2003). Similarly, DCCA and DCA were used to test for differences in species composition between native and old-field transects based on presence–absence of species.

Binomial analyses were used to identify the most likely indicator species for native and old-field ground cover using both the plot and transect data. Plots (excluding those of unknown disturbance history) containing a given species were assigned a binomial variable based on whether they were native or old field. Binomial tables were used to determine if there was a statistically nonrandom (two-sided $\alpha = 0.05$) association of a species with a particular ground-cover type (Sokal and Rohlf 1995). Each occurrence of a species in a plot or transect was considered to have a 0.5 probability of occurring in either native or old-field cover by chance; for example, occurrence of a species in 5 native plots and none in old-field plots would be assigned the probability of 0.031 (nonrandom). Indicators were defined as plants characteristic of and confined predominately to a particular habitat based on the binomial analysis. This assessment was made separately for plots and transects.

Plant species affiliated with one cover type were placed in one of three categories: 1) restricted to one cover type and in a sufficient number of plots or transects (5 or more) to have a statistically significant affinity ($P \leq 0.031$); 2) found predominately in one cover type with a statistically significant affinity; 3) found only in one cover type in 3 or 4 plots or transects ($P = 0.125$ or 0.063, respectively). Category 3 was designed to identify plants with a trend toward a definitive association with one ground-cover type versus the other, which might be revealed with additional sampling. Species for which plot and transect data contradicted one another were not listed as potential indicators.

RESULTS

A total of 232 plant species in 53 families were identified in the study plots, of which six were non-native species (Appendix A). Five families, Poaceae, Fabaceae, Asteraceae, Fagaceae, and Ericaceae, encompassed 58% of the species, and 31 families were represented by a single species. The number of species

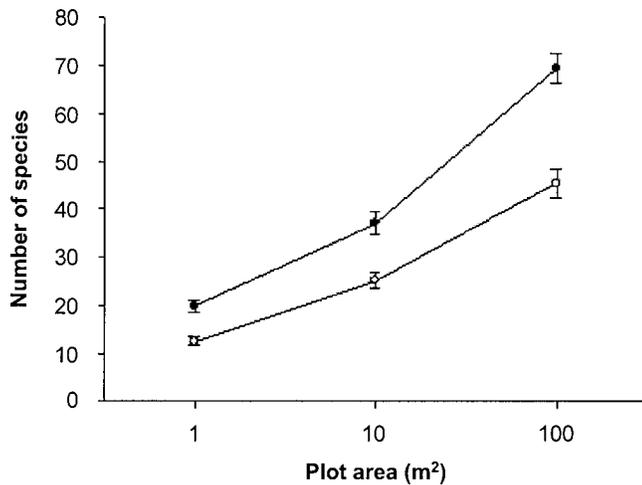


Fig. 1. Species area curve for vegetation plot data collected spring 2003–fall 2005 on Pebble Hill Plantation, Grady and Thomas counties, Georgia. Mean number of species and standard error among 12 plots are given for each scale censused within 100-m² plots. Closed circles indicate native ground cover and open circles indicate old fields.

restricted to one cover type was 56 for native, 15 for old-field, and 16 for unknown-disturbance-history cover plots. Native and old-field plots shared in common 103 species (Appendix A). Average species richness among the 100-m² plots was 69.5 ± 15.9 (\pm SD) for native plots, 45.4 ± 10.8 in old-field plots, and 46.4 ± 10.8 in unknown-disturbance-history plots. Average species richness was approximately 50% higher in native compared to old-field cover at each of the three spatial scales surveyed (Figure 1).

The transect data yielded a total of 432 plant species in 92 families, of which 36 were nonnative species (Appendix A). Five families, Asteraceae, Fabaceae, Poaceae, Cyperaceae, and Fagaceae, contained 49% of the species, and 42 families were represented by a single species. There were 111 species confined to native ground-cover transects and 111 species confined to old-field transects, and 210 species were in both native and old-field transects. Of species confined to native transects, 69% occurred in only 1 of 9 transects, 15% occurred in 2 transects, and 16% occurred in >2 . Of species confined to old fields, 66% occurred in 1 of 8 transects, 29% occurred in 2, and 5% occurred in >2 .

Native and old-field plots were distinctly different in their species composition, as indicated by the Monte Carlo test ($F = 3.827$, $P = 0.002$) and DCA scatter plot (Figure 2). Unknown-disturbance-history plots were largely clustered with old-field plots, although a few appeared to be in a transition zone between old-field and native plots (Figure 2). Similarly, species composition differed significantly between native ground-cover and old-field transects ($F = 1.311$, $P = 0.008$) and appeared to be distinctly different in the scatter plot (Figure 3).

Native sites were characterized by greater dominance of grasses, especially wiregrass and slender bluestem (*Schizachyrium tenerum*), and a lower dominance of forbs and woody species compared to old-field sites (Table 1). Native plots had an average ab-

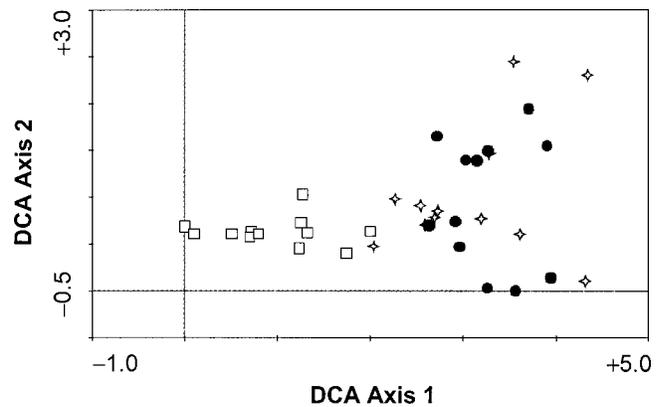


Fig. 2. Ordination diagram of vegetation plot data using detrended correspondence analysis (DCA). Data were collected between spring 2003 and fall 2005 on Pebble Hill Plantation, Grady and Thomas counties, Georgia. Squares indicate native ground-cover plots, circles indicate old-field plots, and stars indicate plots of unknown soil disturbance history. Proximity of symbols reflects their similarity in species composition and cover.

solute cover of 29.6% grasses, 7.0% forbs, and 38.4% woody shrubs and vines. In comparison, old-field sites had an absolute cover of 0.6% grasses, 15.7% forbs, and 63.2% woody shrubs and vines (Table 1). Of the forbs in old-field plots, several species (hyssopleaf thoroughwort [*Eupatorium hyssopifolium*], dogfennel [*E. capillifolium*], lesser snakeroot [*Ageratina aromatica*], and wrinkleleaf goldenrod [*Solidago rugosa*]) are structurally similar to woody shrubs and account for approximately two-thirds of forb cover (Table 1).

Potential indicator species were identified for each cover type by using binomial analyses in both the plot and transect data (Table 2). Twelve species were restrict-

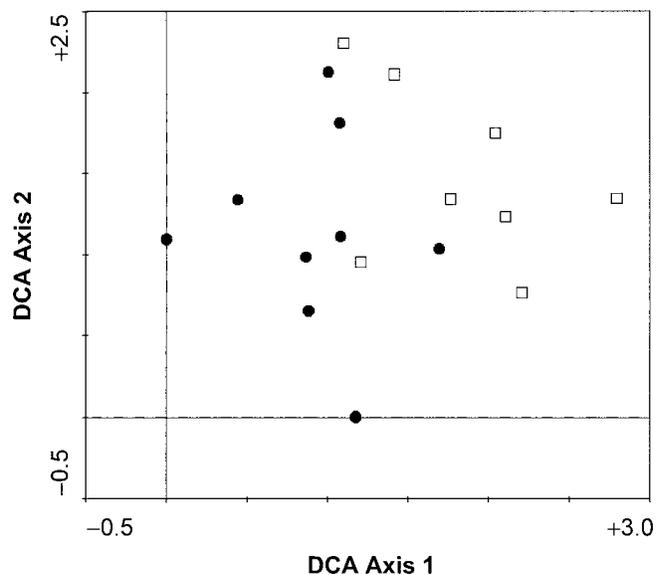


Fig. 3. Ordination diagram of vegetation transect data using detrended correspondence analysis (DCA). Data were collected between spring 2003 and fall 2005 on Pebble Hill Plantation, Grady and Thomas counties, Georgia. Squares indicate native ground-cover transects and circles indicate old-field transects. Proximity of symbols reflects their similarity based on species presence-absence.

Table 1. The 30 most dominant plant species, based on percent absolute cover, for native and old-field plots, in descending order of average cover among study plots, Pebble Hill Plantation, Grady and Thomas counties, Georgia, from spring 2003 to fall 2005.

Native ground-cover sites			Old-field sites		
Species	Growth form ^a	% cover	Species	Growth form ^a	% cover
<i>Aristida stricta</i>	G	24.3	<i>Liquidambar styraciflua</i>	W	14.8
<i>Ilex glabra</i>	W	7.3	<i>Callicarpa americana</i>	W	10.5
<i>Pinus palustris</i>	W	6.2	<i>Pinus taeda</i>	W	7.8
<i>Schizachyrium tenerum</i>	G	4.0	<i>Eupatorium hyssopifolium</i>	F	4.6
<i>Quercus elliotii</i>	W	3.8	<i>Rubus cuneifolius</i>	W	3.9
<i>Quercus incana</i>	W	3.8	<i>Pinus echinata</i>	W	3.6
<i>Quercus laurifolia</i>	W	2.9	<i>Pityopsis aspera</i>	F	3.5
<i>Quercus marilandica</i>	W	2.4	<i>Myrica cerifera</i>	W	2.8
<i>Quercus falcata</i>	W	2.0	<i>Rhus copallinum</i>	W	2.7
<i>Pteridium aquilinum</i>	F	1.9	<i>Quercus falcata</i>	W	2.6
<i>Vaccinium corymbosum</i>	W	1.7	<i>Quercus nigra</i>	W	2.4
<i>Quercus stellata</i>	W	1.2	<i>Prunus serotina</i>	W	2.1
<i>Viola palmata</i>	F	1.1	<i>Ageratina aromatica</i>	F	2.0
<i>Vaccinium darrowii</i>	W	1.0	<i>Eupatorium capillifolium</i>	F	1.8
<i>Rhus copallinum</i>	W	0.9	<i>Quercus stellata</i>	W	1.5
<i>Myrica cerifera</i>	W	0.8	<i>Vitis rotundifolia</i>	V	1.5
<i>Pinus echinata</i>	W	0.8	<i>Erythrina herbacea</i>	F	1.3
<i>Pityopsis aspera</i>	F	0.8	<i>Carya alba</i>	W	1.2
<i>Vaccinium myrsinites</i>	W	0.7	<i>Smilax glauca</i>	V	1.1
<i>Mimosa quadrivalvis</i>	F	0.7	<i>Quercus virginiana</i>	W	1.0
<i>Rhynchosia reniformis</i>	F	0.6	<i>Diospyros virginiana</i>	W	1.0
<i>Sorghastrum nutans</i>	G	0.6	<i>Lespedeza virginica</i>	F	0.9
<i>Dyschoriste oblongifolia</i>	F	0.6	<i>Quercus laurifolia</i>	W	0.7
<i>Gelsemium sempervirens</i>	V	0.6	<i>Rubus argutus</i>	W	0.6
<i>Rubus cuneifolius</i>	W	0.6	<i>Chamaecrista nictitans</i>	F	0.6
<i>Seymeria pectinata</i>	F	0.5	<i>Saccharum alopecuroides</i>	G	0.6
<i>Symphytotrichum concolor</i>	F	0.4	<i>Solidago rugosa</i>	F	0.5
<i>Carya alba</i>	W	0.4	<i>Hypericum hypericoides</i>	F	0.5
<i>Cornus florida</i>	W	0.4	<i>Ampelopsis arborea</i>	V	0.4
<i>Vaccinium arboreum</i>	W	0.4	<i>Parthenocissus quinquefolia</i>	V	0.4

^a Growth form: F, forb; G, grass; V, woody vine; W, woody shrub or tree.

ed to native ground-cover plots with a sufficiently high frequency to be statistically associated with this cover type (category 1), 4 species were statistically associated but were not restricted to native plots (category 2), and 11 species were found only in native cover in 3 or 4 plots (category 3) (Table 2). Old fields had 3 species in category 1, 2 species in category 2, and 0 species in category 3 (Table 2). Native ground-cover transects had 5 indicator species in category 1, 0 species in category 2, and 12 species in category 3. Old-field transects had 0 species in categories 1 or 2 and 8 species in category 3 (Table 2). Considering plots and transects together, a total of 17 species were identified as having statistically significant associations with native ground cover and 5 species had significant associations with old fields (Table 2). An additional 16 species occurring only in native ground cover and 8 species occurring only in old fields had affinities to those cover types, with random probabilities of <0.125.

DISCUSSION

Results from both the study plots and transect data showed distinctive compositional differences between native and old-field sites. In the study plots, this difference was mostly attributable to the larger number of species occurring on native sites and differences in relative cover of species. Old fields were primarily composed of a subset of native species, with only a

few exceptions. Additionally, nonnative species were found in greater numbers in old-field plots and transects than in native ground-cover plots and transects. These results are consistent with other studies comparing old-field and native sites (Means and Grow 1985, Myers 1990, Hedman et al. 2000, Kirkman et al. 2004). Limitations on dispersal and ability for native species to colonize disturbed sites are likely the strongest determinants of species differences between the community types (Kirkman et al. 2004). It should be noted that in our and in the previously cited studies, native pinelands were located on the same properties as the old fields studied, apparently providing a source for colonizing propagules. The proximity of native sites may be a key factor in direction of succession of abandoned old fields (Kirkman et al. 2004).

The plots of unknown soil disturbance history were generally similar to old-field plots in their species composition and dominance, with the exception of some plots that appeared to be in a transition zone between the two. The latter plots may have been subject to various degrees of disturbances other than intensive agriculture, such as those associated with logging or mechanical methods of shrub control. Given the history of selective timber management on our particular study sites, these areas may have been affected to some degree by past vehicular traffic and surface disturbance. The other plots we interpret as being old fields that were abandoned before the earliest available

Table 2. Potential indicator species of native ground cover and old fields from Pebble Hill Plantation, Grady and Thomas counties, Georgia, 2003–2005, and their identification as suggested indicator species in other studies in the southeastern U.S. Coastal Plain. Nomenclature follows Wunderlin and Hansen (2003).

Species	Category ^a		
	Plots	Transects	Other studies ^b
Native ground-cover indicators			
<i>Angelica dentata</i>	3	1	F
<i>Aristida stricta</i>	— ^c	1	B, D, E
<i>Asimina angustifolia</i>	2		
<i>Aureolaria pedicularia</i>	3		
<i>Carphephorus odoratissimus</i>	1	3	
<i>Chrysopsis mariana</i>		1	D, E
<i>Dalea carnea</i>	3		
<i>Dyschoriste oblongifolia</i>	1		B, E
<i>Euphorbia curtisii</i>		3	A
<i>Euphorbia discoidalis</i>	1		A
<i>Gaylussacia dumosa</i>	1	3	
<i>Gaylussacia frondosa</i>	3	1	
<i>Gymnopogon brevifolius</i>	3		
<i>Helianthus angustifolius</i>	3		
<i>Hypericum hypericoides</i>		3	
<i>Ilex glabra</i>	1		
<i>Lobelia amoena</i>		3	
<i>Mimosa quadrivalvis</i>	2		E
<i>Monotropa uniflora</i>	3		
<i>Muhlenbergia capillaris</i>	3	3	E
<i>Panicum virgatum</i>		3	E
<i>Pityopsis graminifolia</i>	1		
<i>Pleopeltis polypodioides</i>		3	
<i>Pteridium aquilinum</i>	1		B, C, E
<i>Quercus elliotii</i>	1		
<i>Quercus incana</i>	1	3	
<i>Quercus laevis</i>	1	3	
<i>Sebastiania fruticosa</i>		3	
<i>Saccharum coarctatum</i>	3		
<i>Salvia azurea</i>	1	1	
<i>Seymeria pectinata</i>		3	
<i>Strophostyles umbellata</i>	2		C, E, F
<i>Stylisma patens</i>	3		
<i>Symphytichum adnatum</i>	1		B, E
<i>Symphytichum concolor</i>	2		C, E
<i>Tephrosia virginiana</i>	3		A, B, C, E, F
Old-field indicators			
<i>Croton glandulosus</i>		3	
<i>Erythrina herbacea</i>	2		
<i>Eupatorium capillifolium</i>	2		
<i>Gamochaeta pensylvanica</i>		3	
<i>Lespedeza procumbens</i>		3	
<i>Liatris tenuifolia</i>		3	
<i>Quercus virginiana</i>	1		
<i>Ruellia caroliniensis</i>		3	A, B, D
<i>Sebastiania fruticosa</i>		3	
<i>Seymeria pectinata</i>		3	
<i>Solidago rugosa</i>	1		
<i>Trichostema dichotomum</i>	1		
<i>Yucca filamentosa</i>		3	

^a Category: 1, plants restricted to one cover type with a statistical affinity; 2, plants predominately in one cover type with a statistical affinity; 3, plants restricted to one cover type without statistical affinity.

^b Other studies: A, Rodgers and Provencher 1999; B, Hedman et al. 2000; C, Dale et al. 2002; D, Smith et al. 2002; E, Kirkman et al. 2004; F, Carter et al. 2004.

^c *Aristida stricta* was used as an indicator of whether or not sites had been disturbed and hence was not included in the analysis.

aerial photograph, based on their close similarity in species composition and structure to confirmed old-field plots. This assessment provides an example of the way in which baseline data characterizing the composition and dominance of native and old-field sites may be used in interpreting the disturbance history of pineland ground cover through multivariate analysis.

In contrast to the study plots, transects had an equal number of species limited to either native or old-field sites. This difference is attributable to the incorporation of a much larger area and thus a greater accumulation of rare species in transects relative to the plots, reflected by the fact that the great majority of species in the transect data occurred in only one or two transects. These species may be either naturally rare species or species associated with isolated disturbances; in either case, they are likely to be missed by transects in the other cover type.

In addition to compositional differences between native and old-field sites, there were significant differences in structure and species dominance. In particular, native sites were characterized by high grass cover and relatively lower forb and woody species cover, whereas old fields had only trace cover by grass and were dominated by large-statured forbs and woody plants. Clearly the likelihood of dispersal and establishment of plants from native areas to abandoned fields is dependent, at least in part, on whether they are grass, forb, or woody species. The association of higher forb and woody species dominance with degree of soil disturbance in southern pinelands has been noted in other studies (Hedman et al. 2000, Dale et al. 2002, Smith et al. 2002). This observation has implications for the management techniques that may be required to restore certain old-field pinelands to a more natural vegetation structure, including more frequent burning (White et al. 1991, Waldrop et al. 1992, Provencher et al. 2001c, Glitzenstein et al. 2003) and use of selective herbicides (Brockway et al. 1998, Litt et al. 2001, Provencher et al. 2001a,b, Jones and Chamberlain 2004, Miller and Miller 2004).

In addition to distinguishing native versus old-field pinelands, our study reveals the potential for old-field pine forests to harbor a significant proportion of native pineland species when managed with frequent fire and selective timber thinning. Such forests are more appropriately described as partially to mostly restored native pinelands rather than purely anthropogenic habitats, as demonstrated by their potential to support a diversity of rare animal species adapted to native pinelands (Masters et al. 2003, Provencher et al. 2003). These qualities attest to the benefits of frequent fire and appropriate timber management for restoring and maintaining high-quality wildlife habitat in post-agriculture pine forests.

We propose 17 species as indicators of native ground cover lacking intensive soil disturbance, based on their statistical affinity to that cover type, and 16 species to be potential indicators based on their nearly significant associations with native sites. Of these, 14 species have been identified as indicators of native ground cover lacking an intensive soil disturbance his-

tory in Coastal Plain pinelands in other studies (Rodgers and Provencher 1999, Hedman et al. 2000, Dale et al. 2002, Smith et al. 2002, Kirkman et al. 2004; Table 2). In addition to wiregrass, used to identify native plots in this study and thus excluded from analysis, coastalplain angelica (*Angelica dentata*), Maryland goldenaster (*Chrysopsis mariana*), oblongleaf snakeherb (*Dyschoriste oblongifolia*), Curtis' spurge (*Euphorbia curtisii*), summer spurge (*Euphorbia discoidalis*), sensitive brier (*Mimosa quadrivalvis*), hairawn muhly (*Muhlenbergia capillaris*), switchgrass (*Panicum virgatum*), brackenfern (*Pteridium aquilinum*), pink fuzzybean (*Strophostyles umbellata*), eastern silver aster (*Symphyotrichum concolor*), and goat's rue (*Tephrosia virginiana*) appear to have geographically broad utility as indicators of pineland sites which have experienced minimal ground-cover disturbance. Other species, live oak (*Quercus virginiana*), Carolina wild petunia (*Ruellia caroliniensis*), wrinkleleaf goldenrod, and forked bluecurls (*Trichostema dichotomum*), appear to be reliable indicators of soil disturbance.

These studies of upland pine forests also point out the variation in floristic composition among pinelands. In individual studies, most of the species identified as potential indicators have not been found in sufficiently large numbers in other studies to merit distinction (Hedman et al. 2000, Dale et al. 2002, Smith et al. 2002, Kirkman et al. 2004). This may be related to inherent site and soil differences (i.e., sandy vs. clayey soils). However, given that it is recommended to use the full suite of plant species available in interpreting natural habitat integrity (Zonneveld 1983, Kremen 1992, Carignan and Villard 2002), all suggested species, when appearing in significant numbers, should be considered as indicative of native community presence. Such interpretations are critical and time sensitive, given the ongoing conversion of native pinelands to other land uses in the southeastern U.S. Coastal Plain.

In summary, southern pineland native ground cover contains a significant number of species that do not readily become reestablished during several years or even decades following extensive soil disturbance. The extreme sensitivity of many native species to intensive soil disturbance underscores the need to identify remaining undisturbed areas for conservation, as facilitated by the use of indicator species suggested by this and other studies. Nevertheless, our data demonstrate the potentially high species richness of native plants in old fields managed with frequent fire and selective timber management. Understanding the differences and similarities between these community types, including species composition and community structure, is important for prioritizing future conservation, restoration, and management of the species they support.

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Appendix A. Plant species list for plots and transects, with number of occurrences for three ground-cover types, Pebble Hill Plantation, Grady and Thomas counties, Georgia, 2003–2005. Nomenclature follows Wunderlin and Hansen (2003). Asterisks denote nonnative species.

Species	Family	Plots ^a			Transects ^a	
		N	O	U	N	O
<i>Acalypha gracilens</i>	Euphorbiaceae	3	2	0	4	5
<i>Acer rubrum</i>	Sapindaceae	2	1	3	3	4
<i>Aesculus pavia</i>	Sapindaceae	0	0	0	0	1
<i>Agalinis divaricata</i>	Orobanchaceae	2	0	0	0	0
<i>Agalinis fasciculata</i>	Orobanchaceae	0	0	0	1	1
<i>Agalinis filifolia</i>	Orobanchaceae	1	0	0	0	0
<i>Agalinis tenuifolia</i>	Orobanchaceae	0	0	0	1	0
<i>Ageratina aromatica</i>	Asteraceae	5	3	8	5	8
<i>Ageratina jucunda</i>	Asteraceae	0	0	0	1	3
<i>Agrimonia microcarpa</i>	Rosaceae	0	0	0	1	0
<i>Albizia julibrissin*</i>	Fabaceae	0	0	0	0	1
<i>Aletris aurea</i>	Nartheciaceae	0	0	0	1	0
<i>Aletris obovata</i>	Nartheciaceae	1	0	0	0	0
<i>Ambrosia artemisiifolia</i>	Asteraceae	0	4	0	4	7
<i>Ampelopsis arborea</i>	Vitaceae	0	3	2	0	1
<i>Amphicarpaea bracteata</i>	Fabaceae	1	0	0	0	0
<i>Andropogon gerardii</i>	Poaceae	2	0	0	0	0
<i>Andropogon gyrans</i>	Poaceae	1	1	0	2	1
<i>Andropogon longiberbis</i>	Poaceae	2	0	0	0	0
<i>Andropogon ternarius</i>	Poaceae	3	0	1	1	1
<i>Andropogon virginicus</i>	Poaceae	2	4	5	2	3
<i>Angelica dentata</i>	Apiaceae	4	0	0	5	0
<i>Anthaenaria villosa</i>	Poaceae	0	0	0	1	0
<i>Apios americana</i>	Fabaceae	0	0	0	0	1
<i>Apocynum cannabinum</i>	Apocynaceae	0	0	0	0	1
<i>Aralia spinosa</i>	Araliaceae	0	0	0	1	3
<i>Arisaema triphyllum</i>	Araceae	0	0	0	0	2
<i>Aristida purpurascens</i>	Poaceae	3	4	2	1	0
<i>Aristida stricta</i>	Poaceae	12	0	3	7	3
<i>Aristolochia serpentaria</i>	Aristolochiaceae	2	0	0	0	0
<i>Arundinaria gigantea</i>	Poaceae	0	0	0	0	2
<i>Asclepias amplexicaulis</i>	Apocynaceae	0	0	0	0	1
<i>Asclepias cinerea</i>	Apocynaceae	1	0	0	0	0
<i>Asclepias tuberosa</i>	Apocynaceae	1	0	1	0	0
<i>Asclepias variegata</i>	Apocynaceae	0	0	0	0	1
<i>Asclepias verticillata</i>	Apocynaceae	0	0	0	2	0
<i>Asimina angustifolia</i>	Annonaceae	8	1	3	6	7
<i>Asimina parviflora</i>	Annonaceae	0	0	0	1	0
<i>Asplenium platyneuron</i>	Aspleniaceae	0	0	0	1	4
<i>Athyrium filix-femina</i>	Dryopteridaceae	0	0	0	0	2
<i>Aureolaria flava</i>	Orobanchaceae	0	0	0	1	2
<i>Aureolaria pedicularia</i>	Orobanchaceae	4	0	0	2	0
<i>Aureolaria virginica</i>	Orobanchaceae	0	0	0	1	0
<i>Baccharis halimifolia</i>	Asteraceae	0	0	0	2	1
<i>Baptisia lecontei</i>	Fabaceae	0	0	0	2	0
<i>Bidens bipinnata</i>	Asteraceae	0	0	0	0	1
<i>Bignonia capreolata</i>	Bignoniaceae	0	0	1	1	4
<i>Boehmeria cylindrica</i>	Urticaceae	0	0	0	0	1
<i>Brickellia eupatorioides</i>	Asteraceae	0	0	0	1	0
<i>Buchnera americana</i>	Orobanchaceae	0	0	0	1	1
<i>Callicarpa americana</i>	Lamiaceae	7	10	12	6	9
<i>Campsis radicans</i>	Bignoniaceae	0	0	0	2	7
<i>Carex abscondita</i>	Cyperaceae	0	0	0	0	1
<i>Carex comosa</i>	Cyperaceae	0	0	0	0	1
<i>Carex retroflexa</i>	Cyperaceae	0	0	0	0	1
<i>Carex verrucosa</i>	Cyperaceae	0	0	0	1	0
<i>Carphephorus corymbosus</i>	Asteraceae	0	0	0	1	0
<i>Carphephorus odoratissimus</i>	Asteraceae	6	0	0	4	0
<i>Carpinus caroliniana</i>	Betulaceae	0	0	0	0	1
<i>Carya alba</i>	Juglandaceae	5	5	7	8	6
<i>Carya glabra</i>	Juglandaceae	3	1	1	2	3
<i>Carya illinoensis*</i>	Juglandaceae	0	0	0	0	2
<i>Castanea pumila</i>	Fagaceae	0	0	1	0	1
<i>Ceanothus americanus</i>	Rhamnaceae	2	0	0	2	3

Appendix A. Continued.

Species	Family	Plots ^a			Transects ^a	
		N	O	U	N	O
<i>Ceanothus microphyllus</i>	Rhamnaceae	0	0	0	2	0
<i>Centrosema virginianum</i>	Fabaceae	4	2	1	4	3
<i>Cercis canadensis</i>	Fabaceae	0	0	0	0	2
<i>Chamaecrista fasciculata</i>	Fabaceae	0	0	0	3	3
<i>Chamaecrista nictitans</i>	Fabaceae	9	10	7	4	2
<i>Chamaesyce hirta</i>	Euphorbiaceae	0	0	0	0	1
<i>Chasmanthium laxum</i>	Poaceae	0	2	4	4	5
<i>Chionanthus virginicus</i>	Oleaceae	0	0	0	1	1
<i>Chrysopsis mariana</i>	Asteraceae	7	2	2	7	0
<i>Cirsium horridulum</i>	Asteraceae	0	0	0	1	0
<i>Clethra alnifolia</i>	Clethraceae	0	0	0	1	0
<i>Clitoria mariana</i>	Fabaceae	6	2	4	3	3
<i>Cnidioscolus stimulosus</i>	Euphorbiaceae	5	3	1	3	6
<i>Commelina erecta</i>	Commelinaceae	0	0	0	1	2
<i>Conoclinium coelestinum</i>	Asteraceae	3	4	0	4	4
<i>Conyza bonariensis</i>	Asteraceae	0	0	0	0	1
<i>Conyza canadensis</i>	Asteraceae	1	1	0	2	2
<i>Cornus florida</i>	Cornaceae	5	2	8	5	7
<i>Crataegus flava</i>	Rosaceae	0	0	0	2	1
<i>Crataegus uniflora</i>	Rosaceae	0	0	1	0	0
<i>Croptilon divaricatum</i>	Asteraceae	0	0	0	1	0
<i>Crotalaria purshii</i>	Fabaceae	0	0	0	2	1
<i>Crotalaria rotundifolia</i>	Fabaceae	3	2	1	2	3
<i>Crotalaria spectabilis*</i>	Fabaceae	0	2	0	0	2
<i>Croton argyranthemus</i>	Euphorbiaceae	0	0	0	0	1
<i>Croton glandulosus</i>	Euphorbiaceae	0	0	0	0	4
<i>Cuphea carthagenensis*</i>	Lythraceae	0	0	0	1	2
<i>Cyclosporum leptophyllum</i>	Apiaceae	0	0	0	0	2
<i>Cynodon dactylon*</i>	Poaceae	0	0	0	1	0
<i>Cyperus croceus</i>	Cyperaceae	0	0	0	1	2
<i>Cyperus filiculmis</i>	Cyperaceae	1	0	0	0	0
<i>Cyperus hystricinus</i>	Cyperaceae	0	0	0	1	0
<i>Cyperus plukenetii</i>	Cyperaceae	1	2	0	2	1
<i>Cyperus retrorsus</i>	Cyperaceae	0	0	0	1	2
<i>Cyrilla racemiflora</i>	Cyrtillaceae	0	0	0	1	1
<i>Dalea carnea</i>	Fabaceae	4	0	1	2	0
<i>Dalea pinnata</i>	Fabaceae	0	0	0	2	2
<i>Decumaria barbara</i>	Hydrangeaceae	0	0	0	0	1
<i>Desmodium ciliare</i>	Fabaceae	5	4	3	1	1
<i>Desmodium floridanum</i>	Fabaceae	1	0	1	1	4
<i>Desmodium glabellum</i>	Fabaceae	1	0	0	0	0
<i>Desmodium laevigatum</i>	Fabaceae	0	2	3	3	5
<i>Desmodium lineatum</i>	Fabaceae	1	1	1	4	1
<i>Desmodium marilandicum</i>	Fabaceae	1	0	0	4	3
<i>Desmodium nudiflorum</i>	Fabaceae	0	0	0	1	1
<i>Desmodium obtusum</i>	Fabaceae	0	0	0	1	0
<i>Desmodium paniculatum</i>	Fabaceae	0	3	1	3	5
<i>Desmodium perplexum</i>	Fabaceae	1	0	0	0	0
<i>Desmodium rotundifolium</i>	Fabaceae	0	0	0	2	0
<i>Desmodium strictum</i>	Fabaceae	1	0	1	1	0
<i>Desmodium tenuifolium</i>	Fabaceae	1	0	0	2	0
<i>Desmodium viridiflorum</i>	Fabaceae	1	0	1	4	4
<i>Dichanthelium aciculare</i>	Poaceae	4	3	1	3	1
<i>Dichanthelium acuminatum</i>	Poaceae	1	1	0	2	1
<i>Dichanthelium boscii</i>	Poaceae	0	0	0	2	5
<i>Dichanthelium commutatum</i>	Poaceae	2	1	6	0	2
<i>Dichanthelium dichotomum</i>	Poaceae	0	0	1	0	0
<i>Dichanthelium ensifolium</i>	Poaceae	3	2	2	1	0
<i>Dichanthelium laxiflorum</i>	Poaceae	0	1	0	0	2
<i>Dichanthelium oligosanthes</i>	Poaceae	1	2	1	0	0
<i>Dichanthelium ovale</i>	Poaceae	5	3	0	0	0
<i>Dichanthelium ravenelii</i>	Poaceae	1	1	1	0	0
<i>Dichanthelium sphaerocarpon</i>	Poaceae	1	4	1	0	0
<i>Dichanthelium strigosum</i>	Poaceae	1	1	0	0	0
<i>Dichondra carolinensis</i>	Convolvulaceae	0	0	0	0	2
<i>Digitaria ciliaris</i>	Poaceae	0	0	0	0	1
<i>Digitaria filiformis</i>	Poaceae	2	4	0	0	0
<i>Diodia teres</i>	Rubiaceae	2	2	1	4	4

Appendix A. Continued.

Species	Family	Plots ^a			Transects ^a	
		N	O	U	N	O
<i>Diodia virginiana</i>	Rubiaceae	0	0	0	3	1
<i>Dioscorea bulbifera</i> *	Dioscoreaceae	0	0	1	0	0
<i>Dioscorea floridana</i>	Dioscoreaceae	0	0	0	1	0
<i>Diospyros virginiana</i>	Ebenaceae	8	8	8	6	9
<i>Drosera brevifolia</i>	Droseraceae	0	0	0	1	0
<i>Duchesnea indica</i> *	Rosaceae	0	0	0	1	0
<i>Dyschoriste oblongifolia</i>	Acanthaceae	11	0	2	5	2
<i>Elaeagnus pungens</i> *	Elaeagnaceae	0	0	0	0	1
<i>Elaeagnus umbellata</i> *	Elaeagnaceae	0	0	0	0	1
<i>Elephantopus carolinianus</i>	Asteraceae	0	0	0	0	1
<i>Elephantopus elatus</i>	Asteraceae	9	5	4	6	3
<i>Elephantopus nudatus</i>	Asteraceae	0	0	0	1	1
<i>Elephantopus tomentosus</i>	Asteraceae	0	0	0	1	1
<i>Eragrostis eliottii</i>	Poaceae	0	1	0	0	0
<i>Eragrostis virginica</i>	Poaceae	0	0	0	1	0
<i>Erythites hieracifolia</i>	Asteraceae	0	0	0	2	4
<i>Erigeron strigosus</i>	Asteraceae	0	0	0	0	2
<i>Eriocaulon decangulare</i>	Eriocaulaceae	0	0	0	1	0
<i>Eryngium yuccifolium</i>	Apiaceae	1	0	0	4	1
<i>Erythrina herbacea</i>	Fabaceae	1	7	1	3	6
<i>Eupatorium album</i>	Asteraceae	4	0	1	5	3
<i>Eupatorium altissimum</i>	Asteraceae	0	0	0	0	1
<i>Eupatorium capillifolium</i>	Asteraceae	1	7	1	1	1
<i>Eupatorium compostifolium</i>	Asteraceae	2	1	2	6	8
<i>Eupatorium hyssopifolium</i>	Asteraceae	4	8	5	4	7
<i>Eupatorium mohrii</i>	Asteraceae	0	0	0	3	1
<i>Eupatorium perfoliatum</i>	Asteraceae	0	0	0	0	1
<i>Eupatorium rotundifolium</i>	Asteraceae	4	2	1	4	2
<i>Eupatorium semiserratum</i>	Asteraceae	0	0	0	2	0
<i>Euphorbia curtisii</i>	Euphorbiaceae	0	0	0	3	0
<i>Euphorbia discoidalis</i>	Euphorbiaceae	6	0	2	0	0
<i>Euphorbia pubentissima</i>	Euphorbiaceae	0	0	0	4	1
<i>Euthamia caroliniana</i>	Asteraceae	1	0	0	0	1
<i>Fagus grandifolia</i>	Fagaceae	0	0	0	0	2
<i>Fraxinus americana</i>	Oleaceae	0	0	1	1	3
<i>Galactia erecta</i>	Fabaceae	1	0	1	1	0
<i>Galactia regularis</i>	Fabaceae	0	0	1	1	1
<i>Galactia villosa</i>	Fabaceae	3	1	2	3	6
<i>Galium pilosum</i>	Rubiaceae	11	7	10	8	7
<i>Gamochaeta americana</i>	Asteraceae	0	0	0	1	0
<i>Gamochaeta falcata</i>	Asteraceae	0	0	0	0	1
<i>Gamochaeta pennsylvanica</i>	Asteraceae	0	1	0	0	3
<i>Gaura angustifolia</i>	Onagraceae	0	0	0	1	0
<i>Gaura filipes</i>	Onagraceae	2	0	0	0	1
<i>Gaylussacia dumosa</i>	Ericaceae	5	0	0	3	0
<i>Gaylussacia frondosa</i>	Ericaceae	4	0	0	4	0
<i>Gelsemium sempervirens</i>	Gelsemiaceae	4	5	12	8	8
<i>Gentiana villosa</i>	Gentianaceae	0	0	0	0	1
<i>Geranium carolinianum</i>	Geraniaceae	0	0	0	0	1
<i>Gratiola pilosa</i>	Veronicaceae	0	0	0	1	0
<i>Gymnopogon ambiguus</i>	Poaceae	4	5	3	4	5
<i>Gymnopogon brevifolius</i>	Poaceae	3	0	0	0	0
<i>Hamamelis virginiana</i>	Hamamelidaceae	0	0	0	1	1
<i>Helianthemum carolinianum</i>	Cistaceae	1	0	0	0	0
<i>Helianthus angustifolius</i>	Asteraceae	3	0	1	2	0
<i>Helianthus radula</i>	Asteraceae	0	0	0	2	0
<i>Heteropogon melanocarpus</i>	Poaceae	0	0	0	1	0
<i>Hexalectris spicata</i>	Orchidaceae	0	0	0	1	0
<i>Hibiscus aculeatus</i>	Malvaceae	0	0	0	3	2
<i>Hieracium gronovii</i>	Asteraceae	6	3	3	4	3
<i>Houstonia procumbens</i>	Rubiaceae	11	5	5	4	3
<i>Hypericum crux-andreae</i>	Clusiaceae	3	2	1	3	0
<i>Hypericum galioides</i>	Clusiaceae	0	0	0	1	1
<i>Hypericum gentianoides</i>	Clusiaceae	0	0	0	2	1
<i>Hypericum hypericoides</i>	Clusiaceae	2	9	6	5	8
<i>Hypericum microsepalum</i>	Clusiaceae	0	0	0	0	1
<i>Hypericum mutilum</i>	Clusiaceae	0	0	0	0	1
<i>Hypericum setosum</i>	Clusiaceae	0	0	0	1	0
<i>Hypericum suffruticosum</i>	Clusiaceae	0	0	0	1	0

Appendix A. Continued.

Species	Family	Plots ^a			Transects ^a	
		N	O	U	N	O
<i>Hypericum tetrapetalum</i>	Clusiaceae	0	0	0	0	1
<i>Hyptis mutabilis</i> *	Lamiaceae	0	0	0	0	2
<i>Hypoxis juncea</i>	Hypoxidaceae	1	0	0	0	0
<i>Ilex cassine</i>	Aquifoliaceae	0	0	0	1	0
<i>Ilex coriacea</i>	Aquifoliaceae	0	0	0	2	1
<i>Ilex glabra</i>	Aquifoliaceae	5	0	1	5	2
<i>Ilex opaca</i>	Aquifoliaceae	0	0	0	1	5
<i>Ilex vomitoria</i>	Aquifoliaceae	0	0	0	0	2
<i>Indigofera caroliniana</i>	Fabaceae	0	0	0	0	1
<i>Ipomoea pandurata</i>	Convolvulaceae	0	0	0	0	1
<i>Ipomoea purpurea</i>	Convolvulaceae	0	0	0	1	0
<i>Itea virginica</i>	Iteaceae	0	0	0	2	1
<i>Jacquemontia tamnifolia</i>	Convolvulaceae	0	0	0	0	2
<i>Juncus coriaceous</i>	Juncaceae	0	0	0	0	2
<i>Juncus repens</i>	Juncaceae	0	0	0	1	0
<i>Juniperus virginiana</i>	Cupressaceae	0	0	0	0	1
<i>Kummerowia striata</i> *	Fabaceae	0	0	0	0	2
<i>Lactuca canadensis</i>	Asteraceae	0	1	0	0	1
<i>Lactuca floridana</i>	Asteraceae	0	1	0	0	0
<i>Lactuca graminifolia</i>	Asteraceae	0	0	0	1	2
<i>Lechea minor</i>	Cistaceae	2	0	0	1	0
<i>Lechea mucronata</i>	Cistaceae	0	0	0	2	4
<i>Lechea pulchella</i>	Cistaceae	1	0	0	0	0
<i>Leersia virginica</i>	Poaceae	0	0	0	0	1
<i>Lespedeza bicolor</i> *	Fabaceae	0	0	1	0	0
<i>Lespedeza capitata</i>	Fabaceae	0	0	0	1	0
<i>Lespedeza cuneata</i> *	Fabaceae	0	0	0	1	0
<i>Lespedeza hirta</i>	Fabaceae	4	3	4	0	2
<i>Lespedeza procumbens</i>	Fabaceae	0	0	1	0	3
<i>Lespedeza repens</i>	Fabaceae	1	1	0	1	2
<i>Lespedeza stuevei</i>	Fabaceae	0	0	1	0	0
<i>Lespedeza violacea</i>	Fabaceae	1	4	1	0	0
<i>Lespedeza virginica</i>	Fabaceae	2	2	4	0	1
<i>Leucothoe racemosa</i>	Ericaceae	0	0	0	1	0
<i>Liatris elegans</i>	Asteraceae	2	3	0	4	0
<i>Liatris gracilis</i>	Asteraceae	0	0	0	3	2
<i>Liatris graminifolia</i>	Asteraceae	0	0	1	2	2
<i>Liatris tenuifolia</i>	Asteraceae	2	0	0	3	0
<i>Ligustrum lucidum</i> *	Oleaceae	0	0	0	0	1
<i>Ligustrum sinense</i> *	Oleaceae	0	0	0	1	1
<i>Linum floridanum</i>	Linaceae	0	0	0	1	0
<i>Liquidambar styraciflua</i>	Altingiaceae	5	11	11	8	9
<i>Liriodendron tulipifera</i>	Magnoliaceae	0	0	0	1	2
<i>Lobelia amoena</i>	Campanulaceae	0	0	0	4	0
<i>Lobelia puberula</i>	Campanulaceae	1	1	1	1	0
<i>Lonicera japonica</i> *	Caprifoliaceae	0	0	0	1	2
<i>Lonicera sempervirens</i>	Caprifoliaceae	0	0	0	1	3
<i>Ludwigia hirtella</i>	Onagraceae	0	0	0	1	0
<i>Ludwigia pilosa</i>	Onagraceae	0	0	0	1	0
<i>Ludwigia virgata</i>	Onagraceae	0	0	0	1	0
<i>Lycopodiella alopecuroides</i>	Lycopodiaceae	1	0	0	0	0
<i>Lygodium japonicum</i> *	Schizaeaceae	0	0	1	4	5
<i>Lyonia lucida</i>	Ericaceae	0	0	0	4	1
<i>Magnolia grandiflora</i>	Magnoliaceae	0	0	0	4	3
<i>Magnolia virginiana</i>	Magnoliaceae	0	0	0	2	3
<i>Malus angustifolia</i>	Rosaceae	0	0	0	4	2
<i>Manfreda virginica</i>	Agavaceae	0	0	0	2	1
<i>Mecardonia acuminata</i>	Veronicaceae	0	0	0	1	0
<i>Melia azedarach</i> *	Meliaceae	0	0	0	0	1
<i>Mikania scandens</i>	Asteraceae	0	0	0	0	1
<i>Mimosa quadrivalvis</i>	Fabaceae	11	3	2	5	4
<i>Mitchella repens</i>	Rubiaceae	0	0	7	2	3
<i>Monarda punctata</i>	Lamiaceae	0	0	0	1	0
<i>Monotropa uniflora</i>	Ericaceae	3	0	0	2	0
<i>Morus rubra</i>	Moraceae	0	0	0	1	1
<i>Muhlenbergia capillaries</i>	Poaceae	4	0	1	3	0
<i>Mycrica carolinensis</i>	Myricaceae	0	0	0	0	1
<i>Myrica cerifera</i>	Myricaceae	10	8	7	5	7
<i>Nyssa sylvatica</i>	Cornaceae	0	0	0	5	6

Appendix A. Continued.

Species	Family	Plots ^a			Transects ^a	
		N	O	U	N	O
<i>Oenothera biennis</i>	Onagraceae	0	0	0	1	2
<i>Oplismenus hirtellus</i>	Poaceae	0	0	0	0	1
<i>Osmanthus americanus</i>	Oleaceae	0	0	0	0	2
<i>Osmunda cinnamomea</i>	Osmundaceae	0	0	0	1	3
<i>Osmunda regalis</i>	Osmundaceae	0	0	0	0	1
<i>Ostrya virginiana</i>	Betulaceae	0	0	0	0	1
<i>Oxalis corniculata</i>	Oxalidaceae	0	2	1	1	4
<i>Oxydendrum arboretum</i>	Ericaceae	0	0	0	2	2
<i>Panicum anceps</i>	Poaceae	4	1	7	2	5
<i>Panicum tenerum</i>	Poaceae	1	0	0	0	0
<i>Panicum verrocosum</i>	Poaceae	0	1	0	0	1
<i>Panicum virgatum</i>	Poaceae	0	0	0	4	0
<i>Parthenocissus quinquefolia</i>	Vitaceae	1	6	5	2	8
<i>Paspalum bifidum</i> *	Poaceae	0	0	1	0	0
<i>Paspalum dilatatum</i> *	Poaceae	0	0	0	1	0
<i>Paspalum floridanum</i>	Poaceae	0	1	0	1	1
<i>Paspalum notatum</i> *	Poaceae	0	0	0	3	5
<i>Paspalum plicatulum</i>	Poaceae	0	0	0	0	1
<i>Paspalum praecox</i>	Poaceae	0	1	0	0	0
<i>Paspalum setaceum</i>	Poaceae	2	3	1	2	1
<i>Paspalum urvillei</i> *	Poaceae	0	0	0	1	1
<i>Passiflora incarnata</i>	Passifloraceae	0	1	0	1	5
<i>Passiflora lutea</i>	Passifloraceae	0	0	0	0	1
<i>Pediomelum canescens</i>	Fabaceae	0	0	0	2	0
<i>Penstemon multiflorus</i>	Veronicaceae	0	0	0	1	0
<i>Persea palustris</i>	Lauraceae	0	0	0	1	1
<i>Phanopyrum gymnocarpon</i>	Poaceae	0	0	0	0	1
<i>Phlox floridana</i>	Polemoniaceae	1	0	0	0	0
<i>Phlox pilosa</i>	Polemoniaceae	0	0	0	1	0
<i>Photinia pyrifolia</i>	Rosaceae	0	1	4	1	0
<i>Phyllanthus tenellus</i> *	Euphorbiaceae	0	0	0	0	2
<i>Phyllanthus urinaria</i> *	Euphorbiaceae	0	0	0	1	2
<i>Phytalis arenicola</i>	Solnaceae	1	0	0	0	0
<i>Phytolacca americana</i>	Phytolaccaceae	0	0	0	1	2
<i>Pinus echinata</i>	Pinaceae	4	7	10	7	8
<i>Pinus elliotii</i>	Pinaceae	0	0	0	0	2
<i>Pinus glabra</i>	Pinaceae	0	0	0	1	1
<i>Pinus palustris</i>	Pinaceae	10	0	2	8	7
<i>Pinus taeda</i>	Pinaceae	5	4	2	6	9
<i>Piptochaetium avenaceum</i>	Poaceae	0	0	0	1	0
<i>Piriyaeta cistoides</i>	Turneraceae	0	0	0	4	6
<i>Pityopsis aspera</i>	Asteraceae	11	7	5	4	2
<i>Pityopsis graminifolia</i>	Asteraceae	5	0	0	3	5
<i>Plantago virginica</i>	Plantaginaceae	0	1	0	0	0
<i>Platanthera ciliaris</i>	Orchidaceae	0	0	0	0	1
<i>Platanthera cristata</i>	Orchidaceae	0	0	0	1	0
<i>Pleopeltis polypodioides</i>	Polypodiaceae	0	0	0	3	0
<i>Pluchea foetida</i>	Asteraceae	0	0	0	1	0
<i>Polygala grandiflora</i>	Polygalaceae	3	0	0	1	4
<i>Polygala incarnata</i>	Polygalaceae	0	0	1	6	6
<i>Polygala nana</i>	Polygalaceae	2	0	3	3	2
<i>Polygala polygama</i>	Polygalaceae	4	2	0	0	2
<i>Polypremum procumbens</i>	Tetrachondraceae	1	4	1	3	4
<i>Polystichum acrostichoides</i>	Dryopteridaceae	0	0	0	0	2
<i>Prenanthes serpentaria</i>	Asteraceae	0	0	0	0	1
<i>Prunus angustifolia</i>	Rosaceae	0	0	0	0	1
<i>Prunus caroliniana</i>	Rosaceae	0	0	0	0	1
<i>Prunus serotina</i>	Rosaceae	8	11	11	8	9
<i>Prunus umbellata</i>	Rosaceae	0	1	0	2	2
<i>Pseudognaphalium obtusifolium</i>	Asteraceae	0	0	0	0	1
<i>Pteridium aquilinum</i>	Dennstaedtiaceae	10	0	4	7	3
<i>Pycnanthemum flexuosum</i>	Lamiaceae	0	0	0	0	1
<i>Pycnanthemum floridanum</i>	Lamiaceae	0	0	0	1	0
<i>Pyrhopappus carolinianus</i>	Asteraceae	0	0	0	0	1
<i>Quercus alba</i>	Fagaceae	0	2	4	4	3
<i>Quercus elliotii</i>	Fagaceae	5	0	1	3	1
<i>Quercus falcata</i>	Fagaceae	8	7	7	5	6

Appendix A. Continued.

Species	Family	Plots ^a			Transects ^a	
		N	O	U	N	O
<i>Quercus incana</i>	Fagaceae	10	1	2	4	0
<i>Quercus laevis</i>	Fagaceae	5	0	0	4	0
<i>Quercus laurifolia</i>	Fagaceae	4	3	8	4	5
<i>Quercus margaretta</i>	Fagaceae	4	3	1	2	1
<i>Quercus marilandica</i>	Fagaceae	1	0	0	0	0
<i>Quercus michauxii</i>	Fagaceae	0	0	0	0	1
<i>Quercus nigra</i>	Fagaceae	3	7	4	4	9
<i>Quercus stellata</i>	Fagaceae	11	7	8	5	3
<i>Quercus velutina</i>	Fagaceae	0	0	0	1	1
<i>Quercus virginiana</i>	Fagaceae	0	6	5	4	6
<i>Rhapidophyllum hystrix</i>	Arecaceae	0	0	0	0	1
<i>Rhexia virginica</i>	Melastomataceae	0	0	0	1	0
<i>Rhododendron canescens</i>	Ericaceae	0	0	0	1	0
<i>Rhus copallinum</i>	Anacardiaceae	12	11	12	7	7
<i>Rhus glabra</i>	Anacardiaceae	0	0	0	1	1
<i>Rhynchosia difformis</i>	Fabaceae	1	0	0	2	1
<i>Rhynchosia reniformis</i>	Fabaceae	11	1	2	4	4
<i>Rhynchosia tomentosa</i>	Fabaceae	1	0	0	3	3
<i>Rhynchospora chalarocephala</i>	Cyperaceae	0	0	0	1	0
<i>Rhynchospora grayi</i>	Cyperaceae	2	0	0	0	1
<i>Rhynchospora harveyi</i>	Cyperaceae	0	0	0	1	0
<i>Rhynchospora intermedia</i>	Cyperaceae	1	0	0	0	0
<i>Rhynchospora miliacea</i>	Cyperaceae	0	0	0	0	2
<i>Rhynchospora plumosa</i>	Cyperaceae	1	0	0	0	0
<i>Richardia scabra</i> *	Rubiaceae	0	0	0	1	1
<i>Rubus argutus</i>	Rosaceae	7	8	9	1	3
<i>Rubus cuneifolius</i>	Rosaceae	8	11	11	8	9
<i>Rubus flagellaris</i>	Rosaceae	0	0	0	4	4
<i>Rubus trivialis</i>	Rosaceae	1	0	0	1	5
<i>Rudbeckia hirta</i>	Asteraceae	1	0	0	4	1
<i>Ruellia caroliniensis</i>	Acanthaceae	1	0	0	0	3
<i>Ruellia ciliosa</i>	Acanthaceae	1	0	0	0	0
<i>Rumex hastatulus</i>	Polygonaceae	0	0	0	0	1
<i>Sabal minor</i>	Arecaceae	0	0	0	1	2
<i>Sabatia angularis</i>	Gentianaceae	0	0	0	0	2
<i>Sabatia calycina</i>	Gentianaceae	0	0	0	0	1
<i>Saccharum alopecuroides</i>	Poaceae	3	10	4	3	5
<i>Saccharum brevibarbe</i>	Poaceae	0	0	0	1	0
<i>Saccharum coarctatum</i>	Poaceae	3	0	0	0	0
<i>Salix humilis</i>	Salicaceae	0	0	0	2	1
<i>Salvia azurea</i>	Lamiaceae	6	0	1	6	0
<i>Salvia lyrata</i>	Lamiaceae	0	0	0	0	2
<i>Sambucus nigra</i>	Adoxaceae	0	0	0	0	2
<i>Sanicula canadensis</i>	Apiaceae	0	0	0	1	2
<i>Sanicula smallii</i>	Apiaceae	0	0	0	0	1
<i>Sassafras albidum</i>	Lauraceae	4	5	1	6	7
<i>Schizachyrium sanguineum</i>	Poaceae	0	0	0	1	0
<i>Schizachyrium scoparium</i>	Poaceae	9	5	3	0	0
<i>Schizachyrium tenerum</i>	Poaceae	2	0	1	1	0
<i>Scirpus cyperinus</i>	Cyperaceae	0	0	0	1	0
<i>Scleria ciliata</i>	Cyperaceae	4	4	0	0	0
<i>Scleria oligantha</i>	Cyperaceae	0	2	0	0	0
<i>Scleria reticularis</i>	Cyperaceae	0	0	0	1	0
<i>Scleria triglomerata</i>	Cyperaceae	0	0	0	1	3
<i>Scutellaria integrifolia</i>	Lamiaceae	0	0	0	2	3
<i>Scutellaria multiglandulosa</i>	Lamiaceae	0	0	0	1	0
<i>Sebastiania fruticosa</i>	Euphorbiaceae	0	0	0	3	0
<i>Senna obtusifolia</i> *	Fabaceae	0	0	0	1	2
<i>Senna occidentalis</i> *	Fabaceae	0	1	0	0	0
<i>Serenoa repens</i>	Arecaceae	0	0	0	1	0
<i>Sericocarpus tortifolius</i>	Asteraceae	10	0	7	7	5
<i>Setaria parviflora</i>	Poaceae	1	0	0	0	0
<i>Setaria pumila</i> *	Poaceae	0	0	0	0	2
<i>Seymeria cassioides</i>	Orobanchaceae	0	0	0	1	0
<i>Seymeria pectinata</i>	Orobanchaceae	2	0	0	3	0
<i>Sideroxylon lanuginosum</i>	Sapotaceae	0	0	1	0	1
<i>Silphium asteriscus</i>	Asteraceae	0	0	0	4	1
<i>Smilax auriculata</i>	Smilacaceae	3	1	3	7	5
<i>Smilax bona-nox</i>	Smilacaceae	8	5	4	7	5

Appendix A. Continued.

Species	Family	Plots ^a			Transects ^a	
		N	O	U	N	O
<i>Smilax glauca</i>	Smilacaceae	4	8	8	8	8
<i>Smilax laurifolia</i>	Smilacaceae	0	2	0	2	0
<i>Smilax pumila</i>	Smilacaceae	0	1	1	3	3
<i>Smilax rotundifolia</i>	Smilacaceae	0	0	0	3	1
<i>Smilax smallii</i>	Smilacaceae	2	5	3	5	7
<i>Smilax tamnoides</i>	Smilacaceae	5	4	7	3	4
<i>Smilax walteri</i>	Smilacaceae	0	0	0	1	0
<i>Solanum carolinense</i>	Solanaceae	0	0	0	1	3
<i>Solidago arguta</i>	Asteraceae	0	0	0	3	3
<i>Solidago auriculata</i>	Asteraceae	0	0	0	0	1
<i>Solidago canadensis</i>	Asteraceae	2	4	3	1	7
<i>Solidago odora</i>	Asteraceae	8	7	7	7	8
<i>Solidago rugosa</i>	Asteraceae	0	5	3	1	4
<i>Solidago stricta</i>	Asteraceae	1	1	1	1	0
<i>Solidago tortifolia</i>	Asteraceae	1	0	0	0	1
<i>Sonchus oleraceus</i>	Asteraceae	0	0	0	0	1
<i>Sorghastrum elliotii</i>	Poaceae	0	0	0	2	1
<i>Sorghastrum nutans</i>	Poaceae	2	1	7	0	0
<i>Sorghastrum secundum</i>	Poaceae	2	7	7	0	0
<i>Sorghum halepense*</i>	Poaceae	0	0	0	1	1
<i>Spiranthes tuberosa</i>	Orchidaceae	0	0	0	0	2
<i>Sporobolus floridanus</i>	Poaceae	0	0	0	1	0
<i>Sporobolus indicus*</i>	Poaceae	0	0	0	1	0
<i>Sporobolus junceus</i>	Poaceae	1	0	0	1	0
<i>Stillingia sylvatica</i>	Euphorbiaceae	2	0	0	4	1
<i>Strophostyles umbellata</i>	Fabaceae	9	4	2	1	0
<i>Stylisma humistrata</i>	Convolvulaceae	0	0	0	1	0
<i>Stylisma patens</i>	Convolvulaceae	3	0	0	1	0
<i>Stylosanthes biflora</i>	Fabaceae	7	1	6	5	4
<i>Symphyotrichum adnatum</i>	Asteraceae	9	0	0	4	2
<i>Symphyotrichum concolor</i>	Asteraceae	11	1	3	5	2
<i>Symphyotrichum dumosum</i>	Asteraceae	5	4	8	5	1
<i>Symphyotrichum patens</i>	Asteraceae	0	0	0	1	1
<i>Symphyotrichum sagittifolium</i>	Asteraceae	0	0	0	4	3
<i>Symphyotrichum undulatum</i>	Asteraceae	0	0	0	1	0
<i>Symplocos tinctoria</i>	Symplocaceae	1	1	3	2	2
<i>Taxodium ascendens</i>	Cupressaceae	0	0	0	1	0
<i>Tephrosia hispida</i>	Fabaceae	1	0	0	0	2
<i>Tephrosia spicata</i>	Fabaceae	2	1	0	2	2
<i>Tephrosia virginiana</i>	Fabaceae	4	0	0	5	2
<i>Tetragonotheca helianthoides</i>	Asteraceae	0	0	0	0	1
<i>Teucrium canadense</i>	Lamiaceae	1	0	0	0	0
<i>Thelypteris kunthii</i>	Thelypteridaceae	0	0	0	0	1
<i>Tillandsia usneoides</i>	Bromeliaceae	0	0	0	5	3
<i>Toxicodendron pubescens</i>	Anacardiaceae	0	0	0	2	1
<i>Toxicodendron radicans</i>	Anacardiaceae	9	6	8	2	5
<i>Toxicodendron vernix</i>	Anacardiaceae	0	0	0	0	1
<i>Tragia smallii</i>	Euphorbiaceae	0	0	0	0	1
<i>Tragia urens</i>	Euphorbiaceae	4	2	1	2	0
<i>Tragia urticifolia</i>	Euphorbiaceae	0	0	0	0	2
<i>Trichostema dichotomum</i>	Lamiaceae	0	6	1	3	3
<i>Trichostema setaceum</i>	Lamiaceae	2	0	0	0	0
<i>Tridens carolinianus</i>	Poaceae	1	0	0	0	0
<i>Tridens flavus</i>	Poaceae	3	7	6	4	2
<i>Utricularia purpurea</i>	Lentibulariaceae	0	0	0	1	0
<i>Vaccinium arboreum</i>	Ericaceae	5	1	6	8	6
<i>Vaccinium corymbosum</i>	Ericaceae	3	0	0	2	1
<i>Vaccinium darrowii</i>	Ericaceae	5	0	4	2	1
<i>Vaccinium myrsinites</i>	Ericaceae	5	1	0	6	2
<i>Vaccinium stamineum</i>	Ericaceae	0	2	6	6	8
<i>Verbesina aristata</i>	Asteraceae	4	0	1	4	4
<i>Verbesina virginica</i>	Asteraceae	0	0	0	1	2
<i>Vernonia angustifolia</i>	Asteraceae	4	1	2	6	6
<i>Viburnum dentatum</i>	Adoxaceae	0	0	0	0	1
<i>Viburnum nudum</i>	Adoxaceae	0	0	0	0	1
<i>Viburnum rufidulum</i>	Adoxaceae	0	0	0	2	3
<i>Viola palmata</i>	Violaceae	1	0	1	2	0
<i>Viola primulifolia</i>	Violaceae	0	0	0	0	2
<i>Viola sororia</i>	Violaceae	0	0	0	0	1

Appendix A. Continued.

Species	Family	Plots ^a			Transects ^a	
		N	O	U	N	O
<i>Viola villosa</i>	Violaceae	0	0	0	2	1
<i>Vitis aestivalis</i>	Vitaceae	0	3	1	1	3
<i>Vitis cinerea</i>	Vitaceae	0	0	0	0	1
<i>Vitis rotundifolia</i>	Vitaceae	2	5	9	7	10
<i>Wahlenbergia marginata</i>	Campanulaceae	0	0	0	1	3
<i>Woodwardia areolata</i>	Blechnaceae	0	0	0	0	1
<i>Woodwardia virginica</i>	Blechnaceae	0	0	0	1	1
<i>Xyris caroliniana</i>	Xyridaceae	0	0	0	1	0
<i>Yucca filamentosa</i>	Agavaceae	0	0	0	0	3

^a Abbreviations: N, native ground cover; O, old-field ground cover; U, unknown soil disturbance history.

EFFECTS OF PRESCRIBED BURNING FREQUENCY ON AVIAN COMMUNITIES IN LONGLEAF PINE FORESTS IN WESTERN LOUISIANA

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ABSTRACT

We examined the effects of four prescribed burning regimes (annual, 2- to 3-y, 4- to 7-y, and unburned) on avian communities in western Louisiana longleaf pine (*Pinus palustris*) stands for two spring (May and June) and two winter (January and February) seasons. Avian species richness and abundance were evaluated by season. During spring, red-cockaded woodpeckers (*Picoides borealis*), Bachman's sparrows (*Aimophila aestivalis*), and pine warblers (*Dendroica pinus*) showed increasing abundance with increasing frequency of fire. Four resident and six Neotropical migrant species increased with decreased frequency of fire ($P \leq 0.05$). During winter, numbers of red-cockaded woodpeckers, pine warblers, and brown-headed nuthatches (*Sitta pusilla*) increased with increased regularity of fires; no temperate-zone migrant species displayed such an increase. Six resident and six temperate-zone migrant species increased with decreasing frequency of fire ($P \leq 0.05$). Although avian species richness and abundance increased with decreasing frequency of fire, species that increased in abundance or were not present in frequently burned stands are not currently in jeopardy. The longleaf pine-grassland forest type occupies <5% of its original area; therefore, this type must be maintained and its extent expanded because it is critical to some avian species that are declining in numbers.

Keywords: avian communities, birds, longleaf pine, *Pinus palustris*, prescribed burning frequency, western Louisiana.

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INTRODUCTION

Longleaf pine (*Pinus palustris*) ecosystems are fire-maintained, subclimax forests (Wahlenburg 1946) that have occurred along the Atlantic and Gulf coastal plains in the southeastern United States for the past 8 million y (Webb 1986). These ecosystems are dependent on fire to limit encroachment of hardwoods that compose most climax communities on the coastal plains (Engstrom et al. 1984).

Prior to European colonization, longleaf pine was the dominant vegetation on as much as 37.2 million ha of uplands in the Southeast (Frost 1993). However, only a small percentage of that area is currently occupied by longleaf pine forests (Platt et al. 1988, Frost 1993, Landers et al. 1995). Exploitation of virgin forests, fire suppression, reforestation to other pine species, and conversion of longleaf pinelands to other land uses have resulted in the loss of as much as 35.6 million ha of longleaf pine-dominated forests (Frost 1993, Landers et al. 1995, Outcalt and Sheffield 1996).

Frequent burning in a longleaf pine forest results in a pine-grassland savannah with little or no hardwood midstory or understory. In pine-dominated stands, however, the impact of burning on avian abundance and composition is species specific; thus, generalizations about correlations between birds and fire are difficult (Bendell 1974, Sousa 1984, Rotenberry et al. 1993). Burning benefits pine-grassland obligate

species; however, species dependent on deciduous canopy vegetation, midstory and understory trees and shrubs, or accumulated litter are disadvantaged (Dickson 1981, Dickson et al. 1993). In recent years, Neotropical migratory birds (NTMBs) have become the focus of concern due to decreasing abundance throughout the breeding ranges of some species (Robbins et al. 1992). Sauer and Droege (1992) analyzed continent-wide Breeding Bird Survey data and found significant declines in some species, particularly among eastern species breeding in forest ecosystems, including those that are dependent on pine-grassland and other early succession habitats (Jackson 1988). Several studies have investigated the impacts of longleaf pine restoration and management on avian communities (Rutledge and Conner 2002, Tucker et al. 2004, Wood et al. 2004). However, no studies have investigated the effects of prescribed burning frequency on abundance and composition of resident and migratory bird communities in longleaf pine forests in western Louisiana. This study investigated the impacts of burning frequency on avian communities in longleaf pine forests dominated by sawtimber size-class trees (>24.5 cm diameter at breast height [DBH], i.e., diameter at 1.37 m above groundline on uphill side of the tree; Stoddard and Stoddard 1987).

STUDY AREA

This study took place on and adjacent to the Fort Polk Military Reservation, approximately 20 km southeast of Leesville, Louisiana (lat 93°15'N, long

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31°00'W). This area was part of the Louisiana Longleaf Pinelands Region of the Gulf Coastal Plain (Nelson and Zillgitt 1969). It was characterized by gently rolling topography with upland areas dominated by longleaf pine in the overstory. The forests were interspersed with perennial and intermittent streams bordered by riparian zones that were dominated by hardwoods (Hamilton and Yurkunas 1987). Overstory trees were 40–50 y old and were of the sawtimber size class.

Six study sites were selected according to previous burning frequency. Two study sites, Bunkhouse East and Forest Service Road West, were in stands on annual burning cycles. Likewise, there were two study sites, Bunkhouse West and Forest Service Road East, in stands on 2- to 3-y burning cycles. These four sites were on U.S. Forest Service land immediately adjacent to the military reservation. Each site was part of a large (>50 ha) stand that had been prescribed burned for numerous years; normally the burns took place in late February or March. The study site on the 4- to 7-y burning cycle, Castor, was on the military reservation. It also was part of a large stand (>50 ha) that had been burned in late fall by an unplanned fire 4 y prior to the initiation of the study. Fire had been excluded from the unburned study site, Airport, for at least 20 y, and probably for the life of the stand. The entire stand was small (<10 ha) and adjacent to urban areas on the reservation.

Both annually burned sites (Bunkhouse East and Forest Service Road West) were open, park-like, pure longleaf pine stands interspersed with small areas of natural longleaf regeneration; there were few snags (<5/ha) on either site. Bunkhouse East had almost no hardwood midstory (i.e., woody plants >2.54 cm DBH with canopies below the general level of the forest canopy) or understory (i.e., woody plants >1.0 m tall but <2.54 cm DBH). Forest Service Road West had a few midstory hardwoods and scattered understory woody vegetation, mostly concentrated in and around a minor drainage that had no distinguishable channel. Both study sites had dense forest floor vegetation (i.e., plants <1.0 m tall) dominated by grasses with scattered herbaceous and woody plants (Laterza 1999). Laterza (1999:31–45) provides detailed descriptions of study site vegetation.

Overstory vegetation and forest floor characteristics of the 2- to 3-y burning cycle stands (Bunkhouse West and Forest Service Road East) were marginally similar to those of the annually burned stands. However, each had more midstory and understory hardwood species and stems per hectare than did the annually burned sites. Bunkhouse West contained a small perennial stream bordered by a 20-m-wide riparian zone with dense hardwood midstory and understory vegetation; two avian transects ran perpendicularly through the riparian zone. Forest Service Road East contained a shallow drainage similar to that on Forest Service Road West. Numerous scattered understory hardwoods and vines were associated with the drainage. One avian transect crossed perpendicularly through the drainage; another ran diagonally along it such that a portion of the drainage area was included

in the avian sampling area. Snags were rare (<2.5/ha) on each study site (Laterza 1999).

Characteristics of the vegetation on the Castor study site, burned on a 4- to 7-y cycle, were very different from those of the more frequently burned study sites. Castor had a greater overstory hardwood component, and the combined numbers of loblolly (*P. taeda*) and shortleaf pines (*P. echinata*) in the overstory were approximately equal to that of longleaf pine. Also, Castor had dense midstory and understory vegetation dominated by hardwood species. Although the forest floor vegetation was dominated by grasses, an increase in woody plants was evident as compared to the more frequently burned sites. The study site contained approximately 16 snags/ha (Laterza 1999).

The unburned Airport study site had the fewest overstory pines; however, diameter of overstory pines was significantly larger than on the more frequently burned sites. Numbers of overstory hardwood species and stems per hectare were similar to those on Castor and higher than on the other sites. Both the midstory and understory were well developed and very dense; each was dominated by hardwood species. Forest floor vegetation was dominated by woody plant species, and vines were prevalent. The site contained approximately 13 snags/ha (Laterza 1999).

METHODS

Sampling Birds

Birds were sampled using the fixed-width strip transect method (Conner and Dickson 1980). Four transects (i.e., replications), each 100 m wide and 200 m long and spaced at least 100 m apart, were established within each study site. The centerline of each transect was marked with plastic flagging along its entire length, and exterior boundaries were marked as needed. The starting and ending points of each transect were recorded using the Global Positioning System.

Bird counts were performed each spring and winter for 2 y. Spring counts were conducted between mid-May and mid-June, 1995 and 1996; winter counts were performed between mid-January and mid-February, 1996 and 1997 (Conner and Dickson 1980). Prior to the beginning of each season, several training sessions were held for participating observers. These sessions allowed the observers to become familiar with the study sites and the local avifauna.

Each study site was sampled nine mornings each season. Each of three observers sampled a pair of sites each morning. For each pair, the study site sampled first was rotated every other morning; beginning points on each site also were alternated. Sampling of the six study sites was performed on a rotational basis such that each observer sampled each site three times per season. To reduce weather and observer bias, birds were sampled only on days when all three observers could participate. Birds were recorded if they were within the transect, flew into or out of the transect (Conner and Dickson 1980), or flew over the transect and appeared to be associated with overstory vegeta-

Table 1. Average numbers of species and individuals per survey (18 surveys) and results of Kruskal–Wallis tests and associated Nemenyi multiple comparisons by season for study sites used for avian sampling on or adjacent to the Fort Polk Military Reservation in western Louisiana, 1995–1997.

Season	Study site burning regime and name ^a					
	Annual		2–3 y		4–7 y Castor	Unburned Airport
	BNKH East	FSRD West	BNKH West	FSRD East		
Spring						
Avg. no. species/survey	5.44a ^b (21)	6.28ab (22)	6.83ab (26)	7.67b (23)	11.06c (34)	11.06c (29)
Avg. no. individuals/survey	12.67a	13.00a	13.33a	12.83a	20.33b	28.72b
Winter						
Avg. no. species/survey	5.78a (27)	6.67ab (26)	7.28ab (28)	7.39ab (31)	8.28b (26)	11.78c (29)
Avg. no. individuals/survey	18.61a	43.06a	27.39a	25.89a	24.67a	48.56b

^a Abbreviations: BNKH, Bunkhouse; FSRD, Forest Service Road.

^b Within rows, values followed by same letter did not differ ($\alpha = 0.05$). Total numbers of species recorded are in parentheses.

tion. Data recorded for each observation included species name and number of individuals.

Data Analysis

Because of differences in riparian zones in the annual and 2- to 3-y burning regime study sites, each site was considered separately rather than as a replicate. Each day was considered as a single sampling unit; thus, sample size was nine per study site per season. For each study site, Mann–Whitney *U*-tests ($\alpha = 0.10$) were used to test for differences in numbers of bird species and individuals between years by season. If no differences were detected, the two years of data for each study site were pooled; thus, sample size became 18 for each season. Numbers of bird species, individuals, and individuals of selected species were compared among study sites by season using Kruskal–Wallis tests; differences were examined with nonparametric Tukey-type multiple comparisons using the Nemenyi test ($\alpha = 0.05$) (Zar 1996). These tests used ranked observation data pooled by study site and season. Therefore, more weight was given to frequency of occurrence than to actual numbers observed; it was thus possible that average numbers of species, individuals, or individuals of a selected species recorded per survey were similar between two or more study sites, yet were statistically different.

The null hypotheses tested were that 1) for each study site, seasonal avian species richness and abundance did not differ between years; and 2) within a season, avian species richness, abundance, and numbers of individuals of selected species did not differ among study sites.

RESULTS

Avian Communities

During 216 avian surveys, 5,362 individuals of 72 species were observed among all study sites. Eighteen species recorded during spring were also recorded during winter. Mann–Whitney *U*-tests comparing seasonal numbers of species and individuals between years by study sites revealed only 2 of 24 comparisons differed; more individuals were recorded in winter 1997 than in

winter 1996 on two sites (annually burned Forest Service Road West and 2- to 3-y burned Bunkhouse West). Therefore, for each study site, data gathered during the two years were pooled by season for further analyses.

Spring Birds

During the spring surveys, 1,822 individuals representing 46 species of birds were recorded (Laterza 1999:47–48). Average numbers of species and individuals recorded per survey increased with decreasing burning frequency. Multiple range tests formed three groups for species richness and two for avian abundance. For both parameters, values for the 4- to 7-y burned Castor and unburned Airport study sites were similar and were higher than values for the more frequently burned study sites (Table 1). The lowest avian richness and abundance values were recorded on the annually burned Bunkhouse East study site; values recorded there were less than half as large as those recorded on the Castor and Airport study sites. Richness and abundance values for the other three frequently burned study sites were only slightly higher than those recorded on Bunkhouse East (Table 1).

Bachman's sparrows (*Aimophila aestivalis*), northern cardinals (*Cardinalis cardinalis*), pine warblers (*Dendroica pinus*), and Carolina wrens (*Thryothorus ludovicianus*) were the most abundant species recorded during spring surveys. Abundances of these species and 16 others were compared among study sites (Table 2). Ten species were classified as residents (Robbins et al. 1983) and nine as Neotropical migrants (Peterjohn et al. 1995). Among resident species, red-cockaded woodpeckers (*Picoides borealis*), Bachman's sparrows, and pine warblers were more abundant on study sites subjected to annual or 2- to 3-y burning regimes than on the less frequently burned sites. Northern cardinals, Carolina wrens, tufted titmice (*Baeolophus bicolor*), and Carolina chickadees (*Parus carolinensis*) were most abundant on the less frequently burned sites. Red-bellied woodpeckers (*Melanerpes carolinus*), blue jays (*Cyanocitta cristata*), and brown-headed nuthatches (*Sitta pusilla*) showed no differences in abundances among burning regimes.

Six species of NTMBs (great crested flycatcher

Table 2. Average numbers of individuals of selected species per survey and results of Kruskal–Wallis tests and associated Nemenyi multiple comparisons among avian study sites on or adjacent to the Fort Polk Military Reservation in western Louisiana during spring (May–June), 1995 and 1996.

Species	Study site burning regime and name ^a					
	Annual		2–3 y		4–7 y	Unburned Airport
	BNKH East	FSRD West	BNKH West	FSRD East	Castor	
Residents						
Red-cockaded woodpecker	0.06 (1) ^{ab}	1.44 (15) ^b	0.94 (11) ^b	0.67 (7) ^a	0.00	0.00
Red-bellied woodpecker	0.00	0.16 (3)	0.06 (1)	0.11 (2)	0.06 (1)	0.44 (5)
Blue jay	0.50 (6)	0.06 (1)	0.61 (5)	0.50 (6)	0.89 (5)	0.61 (6)
Bachman's sparrow	4.50 (18) ^a	3.61 (18) ^{ab}	3.89 (18) ^a	1.78 (15) ^b	0.56 (8) ^c	0.17 (3) ^c
Northern cardinal	0.11 (2) ^a	0.17 (3) ^a	0.89 (11) ^a	0.78 (8) ^a	2.70 (17) ^b	5.11 (17) ^b
Pine warbler	2.61 (17) ^{ab}	3.11 (16) ^a	2.06 (16) ^{ab}	2.61 (17) ^{ab}	1.67 (13) ^{ab}	1.61 (9) ^b
Carolina wren	0.22 (3) ^a	0.33 (5) ^a	0.44 (7) ^a	0.72 (11) ^{ab}	1.72 (16) ^{bc}	4.44 (16) ^c
Brown-headed nuthatch	1.28 (12)	0.44 (4)	1.06 (7)	1.17 (8)	0.61 (8)	0.56 (5)
Tufted titmouse	0.00	0.06 (1) ^a	0.00	0.06 (1) ^a	0.72 (8) ^{ab}	1.67 (13) ^b
Carolina chickadee	0.00	0.11 (1) ^a	0.00	0.50 (4) ^{ab}	0.72 (9) ^b	0.83 (8) ^b
Neotropical migrants						
Great crested flycatcher	0.00	0.11 (2) ^a	0.06 (1) ^a	0.28 (5) ^{ab}	0.61 (5) ^b	0.83 (11) ^b
Chipping sparrow	0.39 (1)	0.39 (4)	0.39 (4)	0.28 (5)	0.00	0.00
Indigo bunting	1.17 (13) ^a	0.67 (9) ^{ab}	0.11 (2) ^b	0.72 (9) ^{ab}	0.83 (9) ^a	0.61 (6) ^{ab}
Summer tanager	0.33 (5) ^a	0.22 (4) ^a	0.17 (3) ^a	0.33 (4) ^a	2.06 (16) ^b	1.17 (14) ^b
Red-eyed vireo	0.00	0.00	0.06 (1) ^a	0.00	0.67 (9) ^b	1.50 (12) ^b
White-eyed vireo	0.17 (1) ^a	0.00	0.11 (2) ^a	0.33 (6) ^{ab}	0.89 (11) ^{bc}	2.61 (17) ^c
Yellow-breasted chat	0.11 (2) ^a	0.39 (4) ^a	0.28 (4) ^a	1.28 (16) ^b	1.72 (15) ^b	0.11 (3) ^a
Hooded warbler	0.00	0.00	0.33 (5) ^{ab}	0.00	0.50 (9) ^{bc}	1.39 (12) ^c
Blue-gray gnatcatcher	0.00	0.00	0.00	0.06 (1) ^a	1.56 (10) ^b	1.22 (11) ^b

^a Abbreviations: BNKH, Bunkhouse; FSRD, Forest Service Road.

^b Within rows, values followed by same letter did not differ ($\alpha = 0.05$). Numbers in parentheses are frequencies of occurrence among 18 surveys.

[*Myiarchus crinitus*], summer tanager [*Piranga rubra*], red-eyed vireo [*Vireo olivaceus*], white-eyed vireo [*Vireo griseus*], hooded warbler [*Wilsonia citrina*], and blue-gray gnatcatcher [*Poliioptila caerulea*]) were more abundant on the less frequently burned Castor and unburned Airport study sites than on the more frequently burned sites (Table 2). Yellow-breasted chat (*Icteria virens*) numbers were higher on Forest Service Road East (2- to 3-y burn) and Castor (4- to 7-y burn) sites than on other sites. Indigo bunting (*Passerina cyanea*) and chipping sparrow (*Spizella passerina*) numbers were not different among study sites.

Winter Birds

During the 1996 and 1997 winter sampling periods, 3,540 individuals of 44 species were observed among all study sites (Laterza 1999:54–55). As with spring birds, average numbers of species and individuals recorded per survey increased with decreasing burning frequency. Multiple range tests distinguished three groups for species richness and two for bird abundance (Table 1). Although average number of individuals per survey on the annually burned Forest Service Road West study site was similar to that on the unburned Airport study site (Table 1), differences between the two sites were significant because of infrequent observations of large flocks of birds on the Forest Service Road West study site. Species of birds that were relatively abundant during winter included the American goldfinch (*Carduelis tristis*), white-throated sparrow (*Zonotrichia albicollis*), dark-eyed junco (*Junco hyemalis*), and pine warbler (Table 3).

There were sufficient data to compare abundances among study sites for 12 resident and 10 temperate-zone migrant species (Table 3).

During winter, red-cockaded woodpeckers, pine warblers, and brown-headed nuthatches were more abundant on the frequently burned stands than on the Castor or Airport study sites (Table 3). Bachman's sparrows showed a similar though nonsignificant trend (Table 3). Red-cockaded woodpecker and pine warbler abundances were similar during spring (Table 2). However, the brown-headed nuthatch responded differently. During spring, there were no differences in numbers recorded among study sites (Table 2) whereas during winter, this species was more abundant in the annually burned stands than in the other stands (Table 3).

Red-bellied woodpeckers, northern cardinals, Carolina wrens, and Carolina chickadees were more abundant on the unburned Airport study site than on the other study sites. Likewise, the tufted titmouse increased in abundance with decreasing burning frequency. There were no differences in numbers of pileated woodpeckers (*Dryocopus pileatus*) and blue jays among study sites; although numbers of American crows (*Corvus brachyrhynchos*) differed among study sites, they did not exhibit a trend across burning regimes (Table 3).

Species commonly found in western Louisiana only during winter (Robbins et al. 1983) were classified as temperate-zone migrants. Five of 10 temperate-zone migrants displayed increasing abundance with decreasing burning frequency (Table 3). The white-

throated sparrow, yellow-rumped warbler (*Dendroica coronata*), and American robin (*Turdus migratorius*) were most abundant on the unburned Airport study site. Similarly, the yellow-bellied sapsucker (*Sphyrapicus varius*), ruby-crowned kinglet (*Regulus calendula*), and hermit thrush (*Catharus guttatus*) were recorded in greatest numbers on the unburned Airport study site; however, abundances of these species on that site were not different from abundances on the Castor site (4- to 7-y burn) (Table 3).

Dark-eyed juncos and cedar waxwings (*Bombycilla cedrorum*) did not exhibit significant trends among the study sites. These species were usually recorded in flocks; thus, there were large but nonsignificant differences in numbers of each among the study sites. Although there were significant differences among study sites in numbers of American goldfinches, there were no differences across burning regimes. The chipping sparrow, a NTMB (Peterjohn et al. 1995) typically found year-round in the western Louisiana region (Robbins et al. 1983), tended to be higher in abundance on more frequently burned sites, but differences were not significant (Table 3). For the purpose of this study, that species was grouped with temperate-zone migrants.

DISCUSSION

Spring Birds

Resident Species

Frequent prescribed burning benefits resident species such as the red-cockaded woodpecker and Bachman's sparrow, both closely associated with southeastern pine-grassland ecosystems (Dunning 1993, Kaufman 1996). The red-cockaded woodpecker inhabits mature, open pine forests, and control of midstory hardwoods is necessary to prevent colony abandonment (Locke et al. 1983, Conner and Rudolph 1989). In this study, no red-cockaded woodpeckers were observed on study sites where hardwoods were well established in the midstory (i.e., Castor [4- to 7-y burn] and Airport [unburned]). Bachman's sparrows also exhibited strong preferences for stands with annual or 2- to 3-y burning regimes and were a common species in these stands during spring. These results corroborate the importance of the pine-grassland savannah habitat type to this species (Plentovich et al. 1998, Tucker et al. 2004), which is undergoing a population decline in the southern United States (Dunning and Watts 1990, Brennan 1991). Frequent prescribed burning in pine forests is critical for this species and thus is an important management tool for increasing population numbers (Dunning 1993).

Frequent use of prescribed fire, however, may eliminate dead trees and thus discriminate against cavity-nesting species (Wood and Niles 1978). Although we did not test between seasons, brown-headed nuthatches, tufted titmice, and Carolina chickadees appeared to be less abundant on the frequently burned sites during spring than during winter. All three species are cavity nesters, and relatively low spring numbers

were probably due to lack of snags and thus cavities for nest sites.

High spring numbers of tufted titmice and Carolina chickadees in the less frequently burned stands were probably due to greater numbers of snags and better developed understories and midstories than in the more frequently burned stands (Conner et al. 1983). In eastern Texas, Carolina chickadees were abundant in areas with open understories (Whiting 1978, Conner et al. 1983) and were positively associated with density of midstory hardwoods (Conner et al. 1983). These differing understory findings suggest that density of midstory hardwoods is more important to Carolina chickadees than is condition of the understory.

Carolina chickadees were most abundant on the unburned Airport study site, which had a well-developed midstory and understory (Jones 1997). In eastern Texas, this species was abundant in areas with open understories and was positively associated with density of midstory hardwoods (Conner et al. 1983). Results of the two studies suggest that density of midstory hardwoods is more important to the Carolina chickadee than condition of the understory.

Also in eastern Texas, abundances of northern cardinals, Carolina wrens, and tufted titmice were positively correlated with increasing shrub-level and hardwood midstory vegetation density (Conner et al. 1983). Results of this study agree with those findings in that all three species declined as frequency of prescribed burning increased. Northern cardinals and Carolina wrens typically nest in understory or midstory vegetation (Kaufman 1996, Halkin and Linville 1999); thus, it is not surprising that reduction or elimination of these vegetation strata with frequent prescribed fire adversely affected abundances of the two species.

Neotropical Migrants

Six NTMB species, the great crested flycatcher, summer tanager, red-eyed vireo, white-eyed vireo, hooded warbler, and blue-gray gnatcatcher, exhibited increased abundance with decreasing burning frequency. Of these, the great crested flycatcher, summer tanager, red-eyed vireo, and blue-gray gnatcatcher are commonly associated with mixed pine-hardwood forests (Hespenheide 1971, Whiting 1978, Kaufman 1996). Increased midstory and overstory hardwood vegetation in the Castor (4- to 7-y burn) and Airport (unburned) stands apparently benefited these species.

Red-eyed vireos were abundant only on the unburned and least frequently burned study sites where midstory and overstory hardwoods were prevalent (Jones 1997). Results of this study support findings that such vegetation is important for this species. In eastern Texas, red-eyed vireos were positively associated with increasing numbers of tree species, vegetation height, percent hardwood saplings, density of large trees, and numbers of shrub species (Conner et al. 1983). Similarly, in Arkansas and Mississippi (James 1971, Wood et al. 2004), red-eyed vireos in pine forests were typically associated with well-devel-

Table 3. Average numbers of individuals of selected species per survey and results of Kruskal–Wallis tests and associated Nemenyi multiple comparisons among avian study sites on or adjacent to the Fort Polk Military Reservation in western Louisiana during winter (January–February), 1996 and 1997.

Species	Study site burning regime and name ^a					
	Annual		2–3 y		4–7 y Castor	Unburned Airport
	BNKH East	FSRD West	BNKH West	FSRD East		
Residents						
Red-cockaded woodpecker	0.94 (8) ^{ab}	1.94 (16) ^b	2.39 (17) ^b	2.06 (16) ^b	0.00	0.00
Pileated woodpecker	0.39 (7)	0.11 (1)	0.06 (1)	0.33 (6)	0.22 (4)	0.28 (4)
Red-bellied woodpecker	0.00	0.00	0.06 (1) ^a	0.06 (1) ^a	0.22 (4) ^a	1.39 (12) ^b
Blue jay	0.11 (2)	0.17 (3)	0.06 (1)	0.06 (1)	0.00	0.94 (7)
American crow	0.66 (3) ^{ab}	0.17 (2) ^a	0.17 (3) ^{ab}	0.39 (5) ^{ab}	1.06 (5) ^{ab}	1.06 (9) ^b
Bachman's sparrow	0.94 (8)	0.00	0.06 (1)	0.89 (8)	0.06 (1)	0.00
Northern cardinal	0.00	0.00	0.44 (5) ^a	0.89 (8) ^{ab}	1.67 (14) ^b	5.11 (17) ^c
Pine warbler	5.06 (17) ^a	4.28 (15) ^{ab}	3.27 (16) ^{bc}	2.83 (14) ^{bc}	3.06 (17) ^b	1.22 (8) ^c
Carolina wren	0.28 (4) ^a	1.22 (9) ^{ab}	1.00 (12) ^{ab}	1.06 (9) ^{ab}	1.33 (13) ^b	3.39 (16) ^c
Brown-headed nuthatch	2.67 (13) ^{ab}	2.11 (16) ^a	1.61 (9) ^{abc}	0.89 (7) ^c	0.94 (10) ^{bc}	0.39 (4) ^c
Tufted titmouse	0.11 (2) ^a	0.17 (2) ^{ab}	0.33 (1) ^{ab}	0.17 (3) ^{abc}	1.00 (9) ^{bc}	0.83 (9) ^c
Carolina chickadee	0.11 (2) ^a	0.22 (4) ^a	0.28 (3) ^a	0.83 (5) ^a	0.83 (4) ^a	1.56 (13) ^b
Temperate-zone migrants						
Yellow-bellied sapsucker	0.11 (2) ^a	0.00	0.11 (2) ^a	0.11 (2) ^a	0.56 (2) ^{9ab}	1.06 (13) ^b
American goldfinch	0.83 (2) ^a	20.22 (9) ^b	4.67 (6) ^{ab}	6.89 (5) ^{ab}	2.39 (5) ^{ab}	1.72 (5) ^{ab}
White-throated sparrow	0.72 (3) ^a	7.67 (10) ^b	3.89 (12) ^b	4.61 (12) ^b	4.67 (13) ^b	17.22 (18) ^c
Chipping sparrow	0.67 (6)	1.39 (8)	0.28 (3)	0.83 (4)	0.39 (4)	0.17 (3)
Dark-eyed junco	2.67 (2)	0.28 (1)	5.44 (4)	1.28 (4)	0.83 (2)	0.00
Cedar waxwing	0.28 (1)	0.00	0.56 (1)	0.00	1.22 (2)	1.11 (3)
Yellow-rumped warbler	0.06 (1) ^a	0.33 (2) ^a	0.39 (4) ^a	0.17 (1) ^a	0.44 (3) ^a	2.28 (13) ^b
Ruby-crowned kinglet	0.06 (1) ^a	0.00	0.11 (2) ^a	0.00	0.28 (4) ^{ab}	1.11 (10) ^b
Hermit thrush	0.00	0.06 (1) ^a	0.28 (5) ^{ab}	0.22 (4) ^a	0.50 (6) ^b	0.94 (11) ^b
American robin	0.56 (2) ^a	0.00	0.22 (2) ^a	0.28 (3) ^a	1.94 (5) ^a	4.72 (14) ^b

^a Abbreviations: BNKH, Bunkhouse; FSRD, Forest Service Road.

^b Within rows, values followed by same letter did not differ ($\alpha = 0.05$). Numbers in parentheses are frequencies of occurrence among 18 surveys.

oped deciduous midstory and overstory trees. White-eyed vireos and hooded warblers are typically associated with dense, shrubby undergrowth (Whiting 1978, Engstrom et al. 1984). Both species were most abundant on Castor and Airport sites, where less frequent application of prescribed fire encouraged dense understory vegetation. Similarly, Conner et al. (1983) found that white-eyed vireos were positively associated with increasing density of shrub stems, foliage density from 0 to 1 m in height, and numbers of shrub species.

Winter Birds

Resident Species

Generally, habitat associations of the red-cockaded woodpecker, blue jay, northern cardinal, pine warbler, Carolina wren, tufted titmouse, and Carolina chickadee during winter were similar to those during spring. Although pileated woodpeckers were more abundant during winter than during spring, relatively low numbers were recorded during both seasons and no trends were evident. The red-bellied woodpecker, which did not display a significant trend among study sites during spring, was more abundant on the unburned Airport study site than on the remaining study sites during winter; the difference between seasons was due to an increase in numbers on the Airport study site during winter. That increase may have been due to expanded visibility resulting from the lack of leaves on decidu-

ous vegetation. Red-bellied woodpeckers are commonly associated with woodlands dominated by deciduous tree species (Shackelford et al. 2000) and, as such, the Airport study site had the most suitable habitat.

Only 19 Bachman's sparrows were recorded during winter; thus, statistical tests showed no differences among study sites. The reason for the lower numbers of Bachman's sparrows during winter surveys is unknown but is likely a result of reduced visibility and therefore detectability of the species; Bachman's sparrows are secretive and difficult to locate during winter (Wilson et al. 1995, Kaufman 1996). Approximately 95% of the individuals recorded during each season were on the annual and 2- to 3-y burning regime study sites. These results support the conclusions of Tucker et al. (2004) that in the longleaf pine–grassland savannah habitat type, a 2- or 3-y burning rotation is critical for Bachman's sparrows during both seasons.

Unlike during spring, when low numbers of brown-headed nuthatches in the more frequently burned stands was possibly due to lack of snags and thus cavities, this species was among the most common in those study sites during winter. Although no significant difference for brown-headed nuthatch abundance was evident during spring, winter data indicated increasing abundance with increased frequency of fire (Tables 2, 3). As brown-headed nuthatches are associated with open, mature pine forests (O'Halloran and Conner 1987, Kaufman 1996), the annual and 2- to

3-y burning regime study sites had the most suitable habitat during both seasons.

Temperate-Zone Migrants

During winter, the yellow-bellied sapsucker, white-throated sparrow, yellow-rumped warbler, ruby-crowned kinglet, hermit thrush, and American robin increased in abundance with decreasing burning frequency. These species commonly winter in forests with hardwood trees and dense undergrowth (Whiting 1978, Kaufman 1996), and frequent prescribed burning may reduce habitat suitability. Yellow-bellied sapsuckers forage by pecking holes into the cambium of trees and feeding on oozing sap and insects attracted to the sap (Bent 1964). During winter, white-throated sparrows commonly forage in dense thickets that provide cover and where seeds are abundant beneath leaf litter (Falls and Kopachena 1994). Yellow-rumped warblers and ruby-crowned kinglets forage in understory vegetation in deciduous forests, feeding on insects and insect larvae (Kaufman 1996). Typically, robins congregate in large flocks in habitats containing berry-producing trees and shrubs during winter (Whiting 1978, Kaufman 1996). American robins were most abundant during winter on the Castor and Airport study sites, which had high densities of a variety of berry-producing plants such as yaupon (*Ilex vomitoria*), privet (*Ligustrum* spp.), American beautyberry (*Callicarpa americana*), and American holly (*Ilex opaca*).

As during spring, winter abundances of chipping sparrows were not different among study sites. During spring, chipping sparrows were not recorded on the Castor and Airport study sites and lack of a trend was probably due to low numbers recorded. Conversely, wintering chipping sparrows were recorded in adequate numbers to statistically analyze. However, they were recorded on all study sites but with variable abundances that lacked a discernable trend.

MANAGEMENT IMPLICATIONS

Prescribed burning is an important tool with which resource managers can accomplish many objectives. Obviously, the frequency at which prescribed fire is applied in a longleaf pine forest will influence the vegetational structure and composition of individual stands. Although frequent fires in longleaf pine stands that lead to pine-grassland savannah may discriminate against bird species that typically inhabit pine and mixed pine-hardwood stands with well-developed understory and/or midstory vegetation, this habitat type is critical to many avian species, particularly pine-grassland obligates.

Longleaf pine was once the dominant vegetation cover type in the southeastern United States. At present, <5% of this forest type remains. Areas currently containing this vegetation type must be maintained in order to provide suitable habitat for the red-cockaded woodpecker, Bachman's sparrow, and many other pine-grassland species. Furthermore, increasing the

area of longleaf pine-grassland savannah through creation and/or restoration efforts is vital to maintaining or increasing populations of these species. In addition to birds, there is evidence that restoration of the pine-grassland savannah habitat type may be beneficial to many small mammals (Masters et al. 1998) which, in turn, could benefit many bird, mammal, and reptile species that depend on such animals as prey.

Finally, riparian zones and mesic sites influence avian species composition in longleaf pine-grassland savannah habitats. In this study, 16 bird species common to the less frequently burned Castor and unburned Airport study sites were recorded on the four frequently burned study sites in vegetation associated with small drainages. Such areas provided habitat for avian species that are not normally associated with the longleaf pine-grassland savannah habitat type. These results suggest that riparian zones and mesic sites containing hardwoods may be extremely beneficial for maintaining a wide array of bird species in frequently burned longleaf pine stands.

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CHARACTERIZING MECHANICAL AND PRESCRIBED FIRE TREATMENTS FOLLOWING CLEAR-CUTTING OF JACK PINE AND SHORT-TERM TREATMENT EFFECTS ON INSECT COMMUNITIES

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ABSTRACT

Effects of post-harvest mechanical treatment, prescribed fire, and mechanical treatment followed by prescribed fire in jack pine (*Pinus banksiana*) were characterized on 0.4-ha treatment areas on the Baraga Plains in the Upper Peninsula of Michigan. The effects of these treatments on ground-dwelling arthropod communities were investigated. Differences in mean basal area and diameter at breast height among treatments prior to harvest were not significant.

Fuel loading assessments showed that prescribed fire reduced smaller size classes (<5 cm diameter) of fuels, although prescribed fire did not significantly reduce total fuel loading. The percentage of closed jack pine cones was lower in burned areas (0% in prescribed fire-only treatments, 9.6% in mechanical treatment followed by prescribed fire) than in non-burned areas (55.1% in untreated areas and 23.4% in mechanical treatment-only areas). Burn coverage was less in areas that had mechanical treatment followed by prescribed fire than in areas that received only prescribed fire.

In the first post-treatment growing season, prescribed fire decreased the taxonomic richness and Shannon–Wiener diversity of ground-dwelling arthropods. Mean pitfall trap catches for ants (Hymenoptera: Formicidae) were significantly higher in prescribed fire-only areas (64.6 individuals/trap) than in untreated control areas (12.1 individuals/trap). Mean pitfall trap catches for leaf beetles (Coleoptera: Chrysomelidae) and plant bugs (Hemiptera: Miridae) were significantly lower in prescribed fire-only areas (0.0 and 0.7 individuals/trap, respectively) compared with untreated controls (0.5 and 8.1 individuals/trap, respectively). Lower numbers of arachnids and ground beetles (Coleoptera: Carabidae) were trapped in areas treated with a combination of fire and mechanical treatments (2.3 and 2.4 individuals/trap, respectively) than in untreated control areas (7.0 and 5.1 individuals/trap, respectively), and lower numbers of wingless long-horned grasshoppers (Orthoptera: Gryllacrididae) were caught in both of the treatments that included fire (0.1 individuals/trap in both treatments) compared with untreated control areas (0.5 individuals/trap). Continued monitoring of these sites will enable management recommendations to be made with respect to impacts on ground-dwelling arthropod communities.

keywords: fuels, ground-dwelling arthropod, jack pine, mechanical treatment, Michigan, *Pinus banksiana*, prescribed fire.

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INTRODUCTION

Jack pine (*Pinus banksiana*) is widely distributed throughout Canada and the northeastern and north-central United States (Rudolph and Laidly 1990), including the well-drained, sandy outwash of the Baraga Plains in the Upper Peninsula of Michigan (Berndt 1988). Fire can be important to the jack pine ecosystem for stand replacement (Chapman 1952, Ahlgren 1959, Chrosiewicz 1959), and for mineral cycling, seed dispersal, and management of forest pests and diseases (Rouse 1986, Despons and Payette 1992). Periodic stand-replacing fires occur naturally in jack pine (Beaufait 1962, McRae 1979, Cayford and McRae 1983), when seeds encased in serotinous cones are released when heated to at least 45°C (Keeley and Zedler 1998). Fire-scarified mineral soils are ideal for jack pine seed germination (Smith et al. 1997), but

seeds can be destroyed by overly intense fires (Chrosiewicz 1959, Smith et al. 1997). Therefore, stand-replacing fires may be more detrimental to jack pine regeneration in some cases than low-intensity prescribed fires, which may be used for slash removal, soil scarification, and seed distribution following harvesting (Beaufait 1962, Chrosiewicz 1970, Cayford and McRae 1983).

The jack pine barrens ecosystem in Michigan's Upper Peninsula has been characterized as fragmented, due to land use and fire exclusion (Pregitzer and Saunders 1999). Clear-cutting on a 50-y rotation is a common management practice for jack pine (Houseman and Anderson 2002), and mechanical treatments are often utilized in post-harvest jack pine management to control logging slash to enhance natural regeneration. Examples of these mechanical treatments include anchor-chaining, manual collection–slash removal, and using a whole-tree cutting system instead of a cut-to-length system (Smith et al. 1997). The natural fire regime in jack pine of the Baraga Plains is a 25- to 100-y interval (Van Wagner 1978, Heinselman 1981, Whit-

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ney 1986, Pregitzer and Saunders 1999). The last major fire on the Baraga Plains was an escaped campfire in the 1980s (M. Nelson, Michigan Department of Natural Resources, personal communication).

In jack pine, high-intensity crown fires help in seed dispersal and do not decrease seed viability (Beaufait 1960). Prescribed burning exposes and scarifies mineral soil, enhancing regeneration (Chrosiewicz 1970), and can be used to decrease slash, prepare mineral seedbeds, and control forest pests in jack pine (Beaufait 1962, McRae 1979). Branch age is an important determinant of serotiny, and differences in serotiny may be caused by the variation in the fire regime (Gauthier et al. 1993, Radeloff et al. 2004). Fire can be used to enhance jack pine regeneration, but fire intensity is often too low in small jack pine slash fires and too high in large, stand-replacing crown fires (Chrosiewicz 1959). Indeed, in a comparison of vegetation changes between burned and unburned areas of post-harvest jack pine in northern Lower Michigan, there was a lack of natural jack pine regeneration in response to fire (Abrams and Dickmann 1984).

The effects of mechanical treatment alone, or with small-scale prescribed fire, may mimic certain aspects of natural wildfire, such as seed release from serotinous cones and exposure of mineral soil. Comparing effects of mechanical treatments (such as anchor chaining) and prescribed fire is useful for forestry in the Upper Great Lakes area, and in other jack pine-dominated systems. However, the short- and long-term ecological costs of these practices on the Baraga Plains are not known. One of these ecological costs could be the reduction in biodiversity of ground-dwelling arthropods. Ground-dwelling arthropods are important, as they typically account for a large proportion of the species composition in a given area (Schowalter 2000, Hanula and Wade 2003), and many ground-dwelling arthropod taxa are sensitive to disturbance. In this small-scale study we examined the short-term impacts of fire and mechanical treatments on ground-dwelling arthropod communities. Characterizing the effects of fire and mechanical treatments on jack pine slash will assist in the post-harvest site-preparation decision-making process for land managers.

The objectives of this project were to 1) characterize the effects of prescribed fire and mechanical treatment on fuel load, closed cone percentage, and burn coverage in a post-harvest jack pine site; and 2) determine the differences in biodiversity of ground-dwelling arthropods resulting from prescribed fire and mechanical treatment in a post-harvest jack pine site.

STUDY AREA

The study area was approximately 20 ha and located in the Michigan Technological University Ford Center Research Forest, Baraga County, Michigan (WGS84: lat 46°38'41.6"N, long 88°31'51.9"W). This area was generally characterized by excessively drained soils and level sandy outwash that was dominated by jack pine and associated vegetation types (Berndt 1988). The study site

was surrounded by land managed for blueberry (*Vaccinium* spp.) production, other softwood stands (mostly comprising red pine [*Pinus resinosa*] and jack pine), and mixed hardwood–softwood stands. Mean tree age prior to harvest was 50 y.

Baseline forest inventory data were collected in May–June 2003 from each of five 0.04-ha plots in each treatment area (see Methods). Data recorded included species and diameter at breast height (DBH) of trees that were within the plot, and the height of every fifth live tree. Basal area (BA) per tree was determined, and mean BA per plot was calculated in square meters per hectare (Avery and Burkhart 1994).

METHODS

Study Design

The study site was divided into 16 treatment units, with four treatments applied randomly in each of four treatment areas so that each treatment was replicated four times. Treatments were untreated (i.e., control), mechanical site preparation by anchor chaining, prescribed fire, and prescribed fire following anchor chaining. Treatment areas were 0.40 ha, but each extended into a 12-m-wide buffer–firebreak. Including the buffer–firebreak, each treatment area was approximately 0.64 ha. Treatments were installed in the study site randomly with the constraint that one treatment area of each treatment type would be located in one of the four treatments abutting a road to the north so that the study site could be used as a demonstration site (Figure 1).

Inside each 0.40-ha treatment area, five circular 0.04-ha plots were established. The corners of the treatment areas and buffers, and the centers of the 0.04-ha plots were marked with pieces of reinforcing rod with numbered aluminum tags. Following harvest and mechanical treatment, the reinforcing rod markers were found with a metal detector, or corners and centers were relocated and marked again.

The entire study site was clear-cut from September to November 2003. Mechanical site preparation was achieved by anchor chaining in May 2004, prior to any pre-fire data collection. Firelines were installed in May 2004, and prescribed fires occurred in July 2004 under the direction of the Michigan Department of Natural Resources (Figure 2).

Ground-Dwelling Arthropods

Pre-harvest pitfall trapping of ground-dwelling arthropods was carried out in June 2003. Following harvest and mechanical treatment, pre-fire trapping was carried out in May–June 2004, and post-fire trapping was carried out in July–August 2004. A transect of five pitfall traps was installed in each treatment area, for a total of 80 pitfall traps. Azimuth and distance of each transect from the center of each treatment area were determined randomly. Traps were installed at 5-m intervals along the same transect (Ward et al. 2001). Traps consisted of two plastic cups with an 8.3-cm-diameter opening and 12.4-

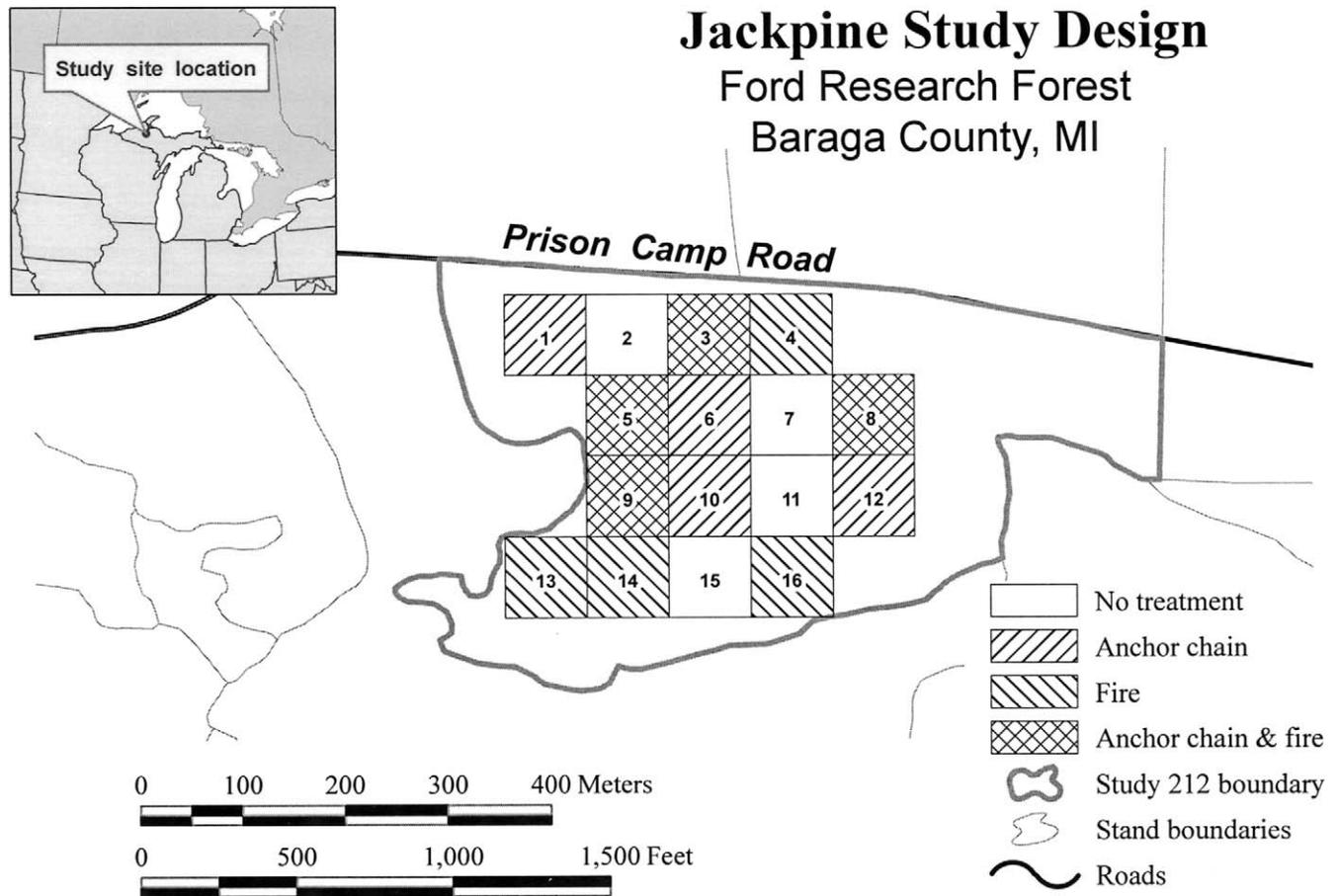


Fig. 1. Treatment installation in a study of the post-harvest effects of mechanical treatment and prescribed fire on jack pine forest biodiversity, Ford Center Research Forest, Baraga County, Michigan. Source: Michael D. Hyslop, Michigan Technological University.

cm depth, one inside the other, inserted into a hole in the ground. A Styrofoam plate was placed above each trap to prevent rainwater from entering. A gap of 2.5 cm was maintained between the plate and the cup by using nails as spacers. The top of the cups in each trap was level with the surface of the forest floor, and each was filled up to 5 cm with a 50% propylene glycol (Prestone® LowTox™ antifreeze) solution. Traps were left open for 1 week, the contents collected, and then the process was repeated, for a total of two rounds of pitfall trap catches per data collection period. Once trap contents were col-



Fig. 2. Aerial photograph of the study site following mechanical treatment and prescribed fire (July 2004) in a study of the post-harvest effects on jack pine forest biodiversity, Ford Center Research Forest, Baraga County, Michigan. Source: Michigan Department of Natural Resources.

lected, insects were counted and identified to family, and arachnids were counted. Data from the five traps in each treatment area at each collection time were pooled.

Fuel Loading

Pre-fire fuel-loading data were collected following mechanical treatment in May 2004 and post-fire fuel-loading data were collected in August 2004. Both fuel loading and closed cone percentage assessments were done using a line-intercept method modified from McRae et al. (1979), which was developed specifically for the jack pine forest type in the Upper Great Lakes area. Fuel loading was assessed by measuring coarse woody debris intersecting fire triangle lines, and classified into one of 24 fuel categories, depending on the size, species, and condition of the fuel. Based on McRae's (1979) procedure, for the first 2 m of a triangle side, all size classes were recorded. For the next 2 m, everything >0.49 cm was recorded. For the next 2 m, everything >0.99 cm was recorded. For the next 2 m, everything >2.99 cm was recorded. For the next 2 m, everything >4.99 cm was recorded. For the last 2 m of a triangle side, everything >7 cm was recorded. Go/no-go gauges made from sheet metal were used to quickly assign fuel to size classes (McRae et al. 1979).

Mean tonnes per hectare was calculated per plot

Table 1. Multiplication factors to determine fuel loading^a in post-harvest jack pine. Modified from McRae et al. (1979).

Fuel diameter (cm)	Fuel type ^b		
	Jack-Red pine	White spruce	Red-Sugar maple
0–0.49	0.0015	0.001	0.00175
0.50–0.99	0.00275	0.00275	0.003
1.00–2.99	0.011	0.01675	0.015
3.00–4.99	0.0395	0.04325	0.052
5.00–6.99	0.08325	0.0825	0.084
>7	0.0015	0.0015	0.002
Rotten	0.001	0.001	0.001

^a To calculate fuel loading in kilograms per square meter, multiply number of intersects of each fuel size and type along triangular transects (each side 12 m long) by the multiplication factor in the table.

^b Jack pine, *Pinus banksiana*; red pine, *Pinus resinosa*; white spruce, *Picea glauca*; red maple, *Acer rubrum*; sugar maple, *Acer saccharum*.

for each size class, as well as for total fuels. Only pieces of fuel whose midpoint intersected the line were counted. Twelve-meter transects were used as fuel triangle sides and final results were then converted from kilograms per hectare to tonnes per hectare (Table 1). Three triangles were randomly located in each treatment area for pre-fire and post-fire assessments, and mean fuel loads per treatment area were calculated. Species results were combined and classified into one of six size categories.

Burn Coverage and Closed Cone Percentage

Percent burn coverage of prescribed fire treatment areas was assessed in August 2004 along six randomly placed 20-m transects into each burned area. The distance (in centimeters) burned or unburned along each transect was measured to calculate mean percent burn coverage.

Pre-fire closed cone percentage data (collected following mechanical treatment in May 2004) and post-fire percentage closed cone (percent serotinous) data (collected in August 2004) were collected by recording all cones attached to woody debris that intersected fuel transect lines in each treatment area. The mean percentage of closed cones was calculated for each treatment area.

Statistical Analyses

For all data collected, means per treatment area at each time of data collection were calculated, with the exception of burn coverage data. For burn coverage data, one-way analysis of variance and least significant difference tests were used to test for differences in mean percent burn coverage between prescribed fire areas and areas treated mechanically prior to prescribed fire. For ground-dwelling arthropod data, taxonomic richness (*S*) and Shannon–Wiener diversity (*H*) were calculated per trap for each treatment area, using taxa that were identified from the trap catches (Hayek and Buzas 1997). Differences among the four treatments were tested using one-way analysis of variance. Arcsine transformations of proportional data

were used for the closed cone percentage assessments and burn coverage assessment, square-root transformations were used for tree diameter data, and log transformations were used for ground-dwelling arthropods in the Formicidae and Arachnida. Where treatment effects were significant, pairwise comparisons between the treatments were made using least significant difference tests. Comparisons of trap catches between dates were not made because of seasonal differences in the data, and data for the two rounds of trapping that occurred post-fire were pooled. Data were analyzed with Statistix 8.0 (Analytical Software 2003). All transformed data were back-transformed for presentation.

RESULTS

Forest Inventory

In the baseline forest inventory, jack pine accounted for 97% of tree species found at the study site. Of the standing jack pines, 19.0% were dead. The mean BA of the study site was 18.22 m²/ha (SE = 1.13), the mean DBH was 19.8 cm (SE = 0.30), and mean tree height was 16.6 m (SE = 0.93). From data collected prior to the harvest, differences in mean BA between treatments were not significant ($F = 1.65$, $df = 3, 12$, $P = 0.230$) nor were differences in mean DBH ($F = 0.38$, $df = 3, 12$, $P = 0.767$).

Fuel Loading

In the pre-fire survey, only the fuel loading in the 0.5–1-cm size class differed significantly between treatments (Figure 3). In the post-fire survey, fuel loading was different among treatments for the four smallest size classes (respectively: $F = 25.73$, $df = 3, 12$, $P < 0.001$; $F = 18.23$, $df = 3, 12$, $P < 0.001$; $F = 17.60$, $df = 3, 12$, $P < 0.001$; $F = 5.58$, $df = 3, 12$, $P = 0.012$). In the post-fire survey, all non-control treatments reduced the fuel loading in the <0.5 cm, 0.5–1-cm, 1–3-cm size classes while fire treatments reduced fuel loading in the 3–5-cm size class (Figure 3). Post-fire differences between treatments in the larger fuel classes and total fuel loading were not significant (Figure 3).

Burn Coverage and Closed Cone Percentage

Burn coverage was 94% in fire-only areas and 61% in areas with mechanical treatment followed by prescribed fire. The difference in mean burn coverage between these two treatments was significant ($F = 11.42$, $df = 1, 6$, $P = 0.015$).

Differences in mean closed cone percentage among treatments were not significant in the pre-fire assessment ($F = 0.76$, $df = 3, 12$, $P = 0.537$). However, differences in mean closed cone percentage among treatments were significant in the post-fire assessment ($F = 30.27$, $df = 3, 12$, $P < 0.001$) (Figure 4). The percentage of closed cones was lowest in fire-only treatments and highest in control areas.

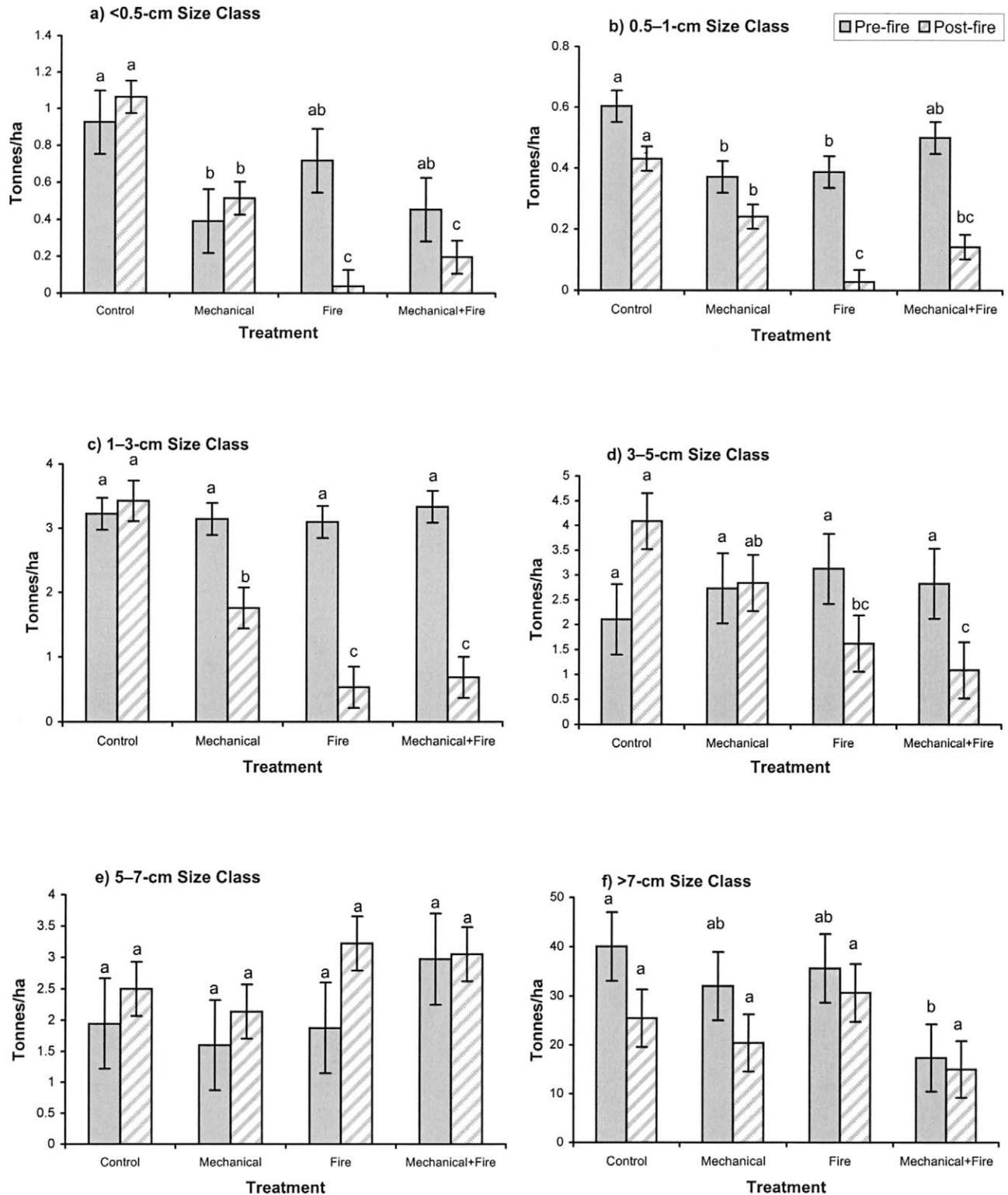


Fig. 3. Pre-fire (May 2004) and post-fire (August 2004) fuel loading by size class in a study of the post-harvest effects of mechanical treatment and prescribed fire on jack pine forest biodiversity, Ford Center Research Forest, Baraga County, Michigan. Bars with different lowercase letters within each data collection period differ significantly.

Ground-Dwelling Arthropods

We identified 22 arthropod taxa that were caught in the 480 pitfall traps installed at the study site. The five most common arthropod taxa in both the pre-har-

vest (Table 2) and post-harvest, pre-fire (Table 3) surveys were Carabidae, Chrysomelidae, Curculionidae, Formicidae, and Arachnida. The five most common arthropod taxa in the post-fire survey were Carabidae, Formicidae, Miridae, Acrididae, and Arachnida (Table

Table 2. Pre-harvest arthropod mean pitfall trap catch (June 2003) in a study of the post-harvest effects of mechanical treatment and prescribed fire on jack pine forest biodiversity, Ford Center Research Forest, Baraga County, Michigan.

Taxon	Treatment ^a				SE
	None	Mechanical	Fire	Mechanical + Fire	
Insecta					
Coleoptera					
Anthicidae	0.00	0.05	0.05	0.05	0.04
Buprestidae	0.00	0.00	0.50	0.00	0.03
Carabidae	2.70a	0.95b	1.80ab	1.70ab	0.44
Cerambycidae	0.00	0.00	0.05	0.00	0.03
Chrysomelidae	1.10	0.85	0.65	0.70	0.25
Curculionidae	0.90	1.00	2.15	3.05	0.80
Elateridae	0.35	0.05	0.40	0.05	0.12
Histeridae	0.00	0.05	0.00	0.00	0.03
Pedilidae	0.00	0.05	0.00	0.00	0.03
Scarabaeidae	0.00	0.05	0.00	0.00	0.03
Scolytidae	0.00	0.05	0.00	0.05	0.04
Staphylinidae	2.60	0.95	1.15	1.65	0.54
Tenebrionidae	0.00	0.00	0.05	0.05	0.04
Hymenoptera					
Formicidae	16.10	11.39	19.84	9.66	0.66
Orthoptera					
Acrididae	0.25	0.40	0.50	0.40	0.19
Tettigoniidae	0.70	0.60	0.65	0.55	0.32
Arachnida					
Acarina, Araneida, and Phalangida	48.09	32.30	45.84	45.51	0.18

^a Results from one-way analysis of variance, with SE ($\alpha = 0.05$). For each order or family, treatment types with different letters differ significantly. Pairwise comparisons between the treatments were made using least significant difference tests (Statistix 8.0; Analytical Software 2003). Data for Formicidae and Arachnida were log-transformed to meet the assumption of homogeneity of variance.

4). In the post-fire arthropod trapping period, more ants (Hymenoptera: Formicidae) and fewer leaf beetles (Coleoptera: Chrysomelidae) and plant bugs (Hemiptera: Miridae) were trapped in the fire-only treatment areas compared with untreated controls. Lower numbers of arachnids and ground beetles (Coleoptera: Carabidae) were trapped in the areas treated with a combination of fire and mechanical treatments than in untreated control treatment areas, and lower numbers of wingless long-horned grasshoppers (Orthoptera: Gryllacrididae) were caught in both of the treatments that included fire compared with the untreated control treatment areas (Table 4).

Differences in taxonomic richness among treatments were not significant in both the pre-harvest ($F = 0.44$, $df = 3, 12$, $P = 0.729$) and pre-fire ($F = 3.33$, $df = 3, 12$, $P = 0.057$) trapping periods. However, differences in taxonomic richness among treatments were significant in the post-fire trapping period ($F = 3.80$, $df = 3, 12$, $P = 0.040$) (Figure 5). Taxonomic richness was lowest in fire-only treatment areas compared with control and mechanical-only treatment areas.

Differences in Shannon–Wiener diversity among treatments were not significant in both the pre-harvest ($F = 0.01$, $df = 3, 12$, $P = 0.999$) and pre-fire ($F = 0.39$, $df = 3, 12$, $P = 0.761$) trapping periods. However, differences in Shannon–Wiener diversity among treatments were significant in the post-fire trapping period ($F = 4.62$, $df = 3, 12$, $P = 0.023$) (Figure 5), being lower in fire-only treatment areas than in all other treatments.

DISCUSSION

The fuel load data, cone serotiny data, and burn coverage data suggest that the prescribed fire treatments and the mechanical treatments followed by fire had sufficient fire intensity to reduce smaller size classes of slash, expose bare mineral soil, and open many of the closed cones in the slash. Post-fire fuel loading differed between treatments in the four smallest size classes but was not significantly different in the largest two size classes. Because prescribed fire had a significant effect on smaller size classes only, total fuel loading was not affected. The smaller size classes were likely consumed rapidly, whereas the larger fuels, although charred, were not entirely consumed.

Over time, jack pine cones naturally open when exposed to heat, so cones that were more exposed to sunlight were more likely to open naturally (Keeley and Zedler 1998). As expected, there was no difference in closed cone percentage between treatments prior to implementation of prescribed fire treatments. The closed cone percentage in the non-burn treatments declined during the study, probably as a result of exposure to sunlight. The heat from the prescribed fire either opened or consumed cones. Percentage of closed cones in all non-control treatments declined during the study, and the percentage of closed cones in the fire-only treatment areas approached zero.

The percentage of area burned was higher in fire-only treatment areas than areas mechanically treated prior to the prescribed fire. Mechanical treatment created more piles of larger fuels, whereas fuels were

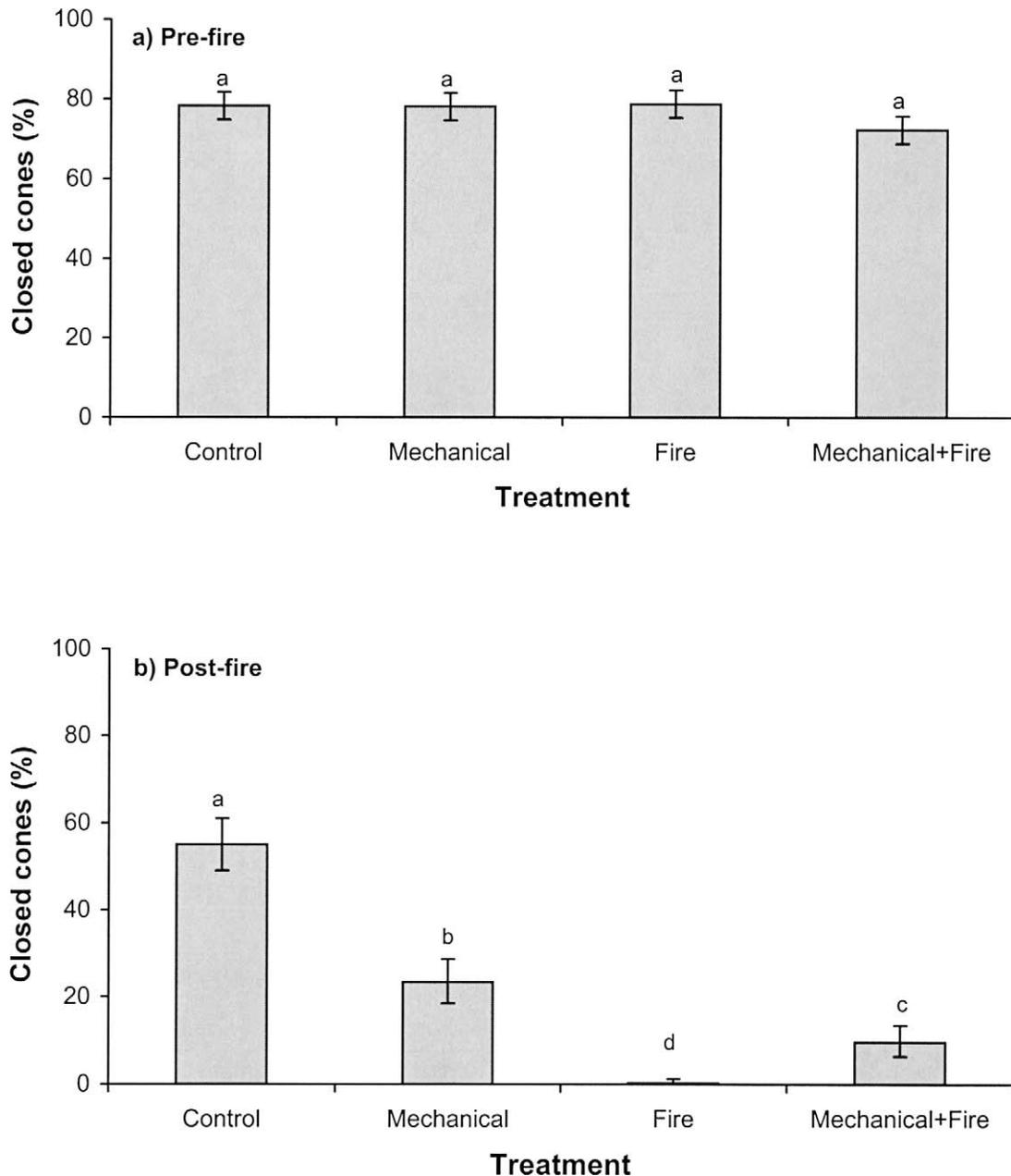


Fig. 4. Jack pine closed cone percentage (a) pre-fire (May 2004) and (b) post-fire (August 2004) in a study of the post-harvest effects of mechanical treatment and prescribed fire on jack pine forest biodiversity, Ford Center Research Forest, Baraga County, Michigan.

more evenly distributed in areas that did not receive mechanical treatment. That even distribution of fuel allowed the fire to carry more completely in fire-only areas than in areas mechanically treated prior to the prescribed fire. Also, soil and slash compaction by heavy machinery was more likely to occur in mechanical-treatment areas than in fire-only areas. By compacting fuels, less air space was available and fuels were not as likely to carry fire. In fire-only treatment areas, more evenly distributed, less compacted fuels allowed for more air space and likely allowed for a more intense burn (Pyne et al. 1996).

We anticipate that the burn intensity will be sufficient to regenerate jack pine in the areas that were burned because of the extent of cone opening and the

coverage of the burn. Other studies have suggested that fire intensity is often too low in jack pine slash and too high in large, stand-replacing crown fires to achieve desired levels of regeneration (Chrosiewicz 1959), and in northern Lower Michigan, a designed post-harvest burn yielded only one plot out of 20 that was considered to have adequate jack pine regeneration (Abrams and Dickmann 1984). Future surveys of our treatment areas will enable us to evaluate the success of the treatments for jack pine regeneration.

Post-fire ground-dwelling arthropod taxonomic diversity and Shannon–Wiener diversity in fire areas decreased, due to the initial impact of fire. However, it remains to be seen whether or not fire eventually produces areas with higher arthropod diversity than me-

Table 3. Pre-fire arthropod mean pitfall trap catch (June 2004) in response to post-harvest effects of mechanical treatment and prescribed fire on jack pine forest biodiversity, Ford Center Research Forest, Baraga County, Michigan.

Taxon	Treatment ^a				SE
	None	Mechanical	Fire	Mechanical + Fire	
Insecta					
Coleoptera					
Anthicidae	0.00	0.00	0.10	0.10	0.05
Buprestidae	0.00	0.00	0.00	0.05	0.03
Carabidae	0.55	0.80	0.25	0.55	0.22
Cerambycidae	0.00	0.00	0.05	0.00	0.03
Chrysomelidae	0.90	1.50	0.90	1.00	0.30
Cleridae	0.00b	0.00b	0.00b	0.10a	0.03
Curculionidae	1.05	1.85	1.80	1.20	0.51
Elateridae	0.00	0.00	0.05	0.00	0.03
Pedilidae	0.00b	0.00b	0.00b	0.10a	0.03
Scarabaeidae	0.15	0.25	0.30	0.35	0.13
Scolytidae	0.05	0.10	0.25	0.20	0.08
Staphylinidae	0.20	0.30	0.35	0.30	0.14
Hymenoptera					
Formicidae	9.62	11.83	3.82	11.96	0.49
Hemiptera					
Miridae	0.10	0.00	0.00	0.00	0.05
Orthoptera					
Acrididae	0.45	0.30	0.20	0.35	0.12
Tettigoniidae	0.15a	0.00b	0.00b	0.00b	0.05
Arachnida					
Acarina, Araneida, and Phalangida	38.99	28.81	30.04	27.79	0.51

^a Results from one-way analysis of variance, with SE ($\alpha = 0.05$). For each order or family, treatment types with different letters differ significantly. Pairwise comparisons between the treatments were made using least significant difference tests (Statistix 8.0; Analytical Software 2003). Data for Formicidae and Arachnida were log-transformed to meet the assumption of homogeneity of variance.

Table 4. Post-fire arthropod mean trap catch (August 2004) in response to the post-harvest effects of mechanical treatment and prescribed fire on jack pine forest biodiversity, Ford Center Research Forest, Baraga County, Michigan.

Taxon	Treatment type ^a				SE
	None	Mechanical	Fire	Mechanical + Fire	
Insecta					
Coleoptera					
Anthicidae	0.30	0.30	0.40	0.55	0.14
Buprestidae	0.05	0.05	0.65	0.60	0.22
Carabidae	5.05a	4.60ab	3.20ab	2.35b	0.79
Cerambycidae	0.10	0.00	0.00	0.05	0.04
Chrysomelidae	0.45a	0.35a	0.00b	0.30ab	0.11
Cleridae	0.00	0.05	0.00	0.00	0.03
Coccinellidae	0.00	0.00	0.00	0.05	0.03
Curculionidae	0.40	0.85	0.35	0.25	0.26
Elateridae	0.05ab	0.00b	0.15b	0.00a	0.04
Pedilidae	0.65	1.25	0.05	0.05	0.69
Scolytidae	0.05	0.05	0.05	0.05	0.05
Staphylinidae	0.25ab	0.45a	0.00b	0.10ab	0.12
Tenebrionidae	0.10	0.20	0.05	0.15	0.12
Hymenoptera					
Formicidae	12.05b	17.78ab	64.60a	14.60ab	0.66
Hemiptera					
Miridae	8.10a	6.80a	0.65b	3.95ab	1.43
Orthoptera					
Acrididae	8.95	9.95	3.35	5.15	2.37
Gryllacrididae	0.50a	0.30ab	0.05b	0.05b	0.11
Gryllidae	0.10	0.45	0.15	0.30	0.13
Tettigoniidae	0.25	0.05	0.05	0.05	0.10
Arachnida					
Acarina, Araneida, and Phalangida	7.03a	5.56ab	3.51ab	2.29b	0.33

^a Results from one-way analysis of variance, with SE ($\alpha = 0.05$). For each order or family, treatment types with different letters differ significantly. Pairwise comparisons between the treatments were made using least significant difference tests (Statistix 8.0; Analytical Software 2003). Data for Formicidae and Arachnida were log-transformed to meet the assumption of homogeneity of variance.

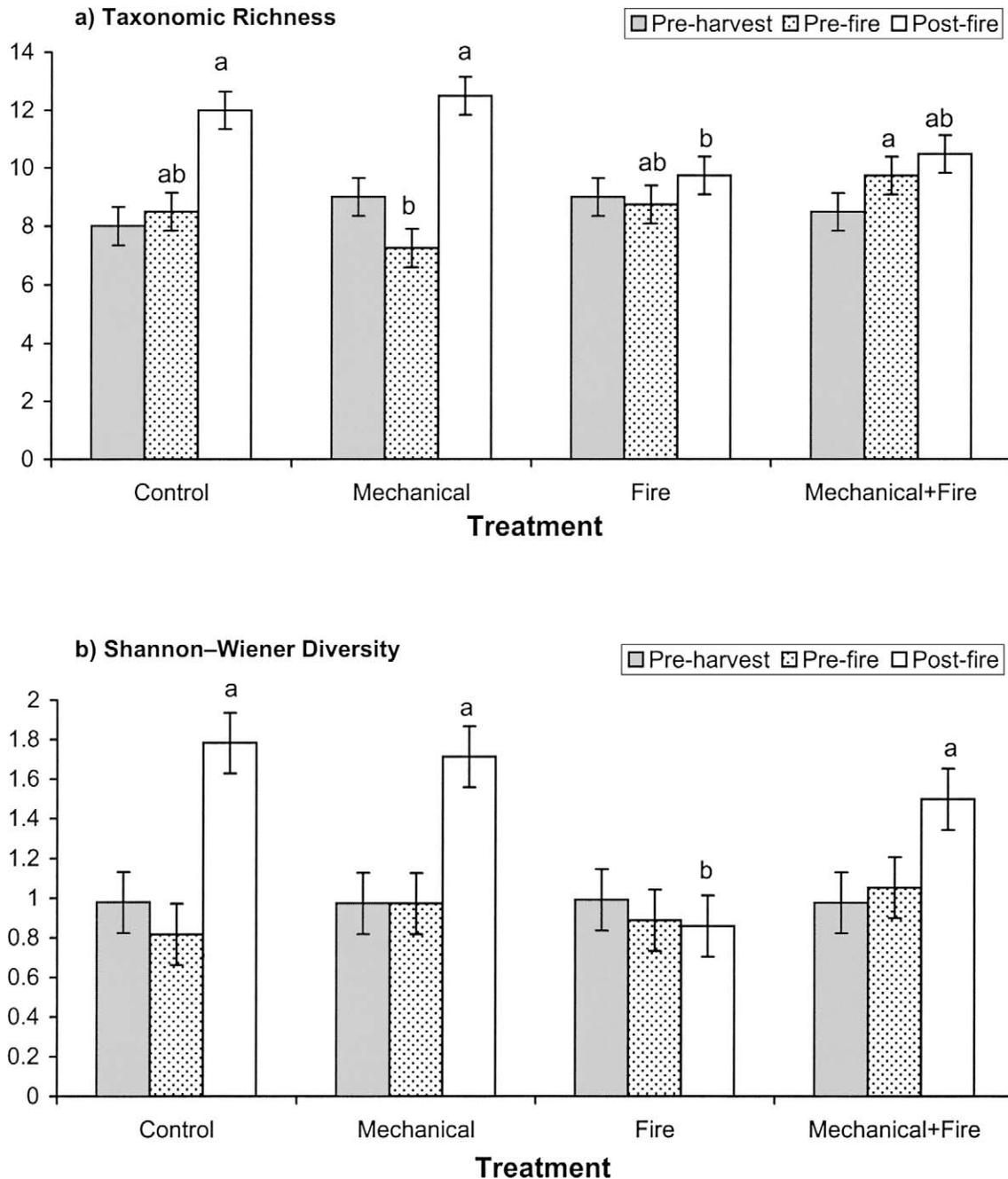


Fig. 5. Pre-harvest (June 2003), pre-fire (June 2004), and post-fire (August 2004) arthropod (a) taxonomic richness and (b) Shannon-Wiener diversity in a study of the post-harvest effects of mechanical treatment and prescribed fire on jack pine forest biodiversity, Ford Center Research Forest, Baraga County, Michigan.

chanically treated or untreated areas. Prescribed fire alone initially decreased arthropod diversity, but further sampling is likely to reveal increased arthropod diversity within 1–2 y as has been documented in prairies (Panzer 2002). Previous studies have indicated that a patchwork of prescribed fires and unburned areas likely results in complete recovery of pre-fire species and could increase abundance of certain species (Beaudry et al. 1997).

Mechanical treatment alone of the study site did not seem to have as much of an effect on arthropod communities as prescribed fire did. Orthopterans were

generally more prevalent in non-fire areas than in treatments that included prescribed fire. In other systems, arachnids tend to be more sensitive to fire than are other arthropod taxa (Andersen and Muller 2000), and arachnids decreased in all areas in our study following prescribed fire treatment. Formicidae (ants) are less sensitive to fire than are other arthropod taxa (Andersen and Muller 2000) and increased in fire-only areas following treatment in this study (Table 4).

Insect responses to prescribed fire are often taxon specific (Beaudry et al. 1997) as they were found to be in this study. For example, population sizes of some

carabid species increase following prescribed burning in jack pine, while others decrease in abundance (Beaudry et al. 1997). Overall carabid trap catch declined in mechanical treatment followed by prescribed fire treatments in our study, though species determinations were not made. Seasonal differences in arthropod abundance probably had a large effect, as data collection occurred at different periods during the summer. Given the normally late start of warm weather on the Baraga Plains, higher arthropod trap catches should normally occur later in the summer regardless of treatment type. As mechanical treatment was performed early in the summer, it is likely that arthropods were not yet fully active and therefore were not as impacted by mechanical treatment as they were by the fire treatment. Since the fire treatment was applied later in the summer than the mechanical treatment, it is likely that more arthropods were either initially killed by the treatment or that it had more of an initial effect on arthropod habitat than the mechanical treatment did (Panzer 2002, Hanula and Wade 2003). Some insect groups were also likely able to escape from the small treatment areas and subsequently recolonize those areas.

The effects that we have documented here are very short-term post-treatment effects, and will likely change over time as the vegetation changes following the treatments. The data presented here reflect short-term effects of the treatments on ground-dwelling arthropod communities. Continued monitoring of long-term treatment effects will be necessary to fully evaluate the potential ecological costs of the fire and mechanical treatments on these communities and to incorporate this information into management recommendations.

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AN ECOSYSTEM APPROACH TO DETERMINING EFFECTS OF PRESCRIBED FIRE ON SOUTHWESTERN BORDERLANDS OAK SAVANNAS: A BASELINE STUDY

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ABSTRACT

Many traditional land management activities and supporting research have concentrated on one or two resources, with limited evaluations of interactions among other potential values. An ecosystem approach to land management requires an evaluation of the blend of physical and biological factors needed to assure productive, healthy ecosystems. Ideally, social and economic values also should be considered. The U.S. Department of Agriculture Forest Service's Coronado National Forest and its partners have just completed a draft for the Peloncillo Programmatic Fire Plan to address fire management strategies for this mountain range, which lies along the southern Arizona–New Mexico border. The plan was designed to reintroduce prescribed or managed fires into an area where wildfires have been excluded since the late 19th century. One persistent question concerned the impacts of cool-season (November–April) and warm-season (May–October) fires on the oak (*Quercus* spp.) ecosystems that are common throughout these mountains. Fires normally occur in June or early July before the summer monsoon rains. However, hotter warm-season fires could damage important wildlife habitats by killing larger numbers of standing trees and shrubs used as nesting sites or cover or as sources of food, and thus some managers prefer burning during the cooler season.

The Rocky Mountain Research Station's Southwestern Borderlands Ecosystem Management Project and its cooperators have initiated a research program to evaluate the impacts of season of burning on a large number of ecosystem components, including hydrology, sedimentation, vegetation, soil nutrient dynamics, small and large mammals, birds, and snakes and other reptiles. Our research is concentrated on 12 small, gauged watersheds that support oak savannas or open woodlands. We plan to burn four watersheds in the warm season, burn four in the cool season, and leave four as controls. The watershed and companion studies are currently in the pre-burn calibration phase. However, little is actually known about the oak ecosystems of the southwestern United States and northern Mexico, where most of these oak stands are found. The preliminary results have provided important new information about these lands. This paper describes the studies and initial results obtained during the pre-treatment phase of this project.

keywords: ecosystem approach, oak (*Quercus*) savannas, prescribed fire, seasonal burning, southwestern Borderlands.

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INTRODUCTION

Fire, which normally occurred as the result of lightning during the late spring and early summer, was the most important natural disturbance in the oak (*Quercus* spp.) woodlands and savannas of the southwestern United States and northern Mexico prior to European–American settlement. Some fires were set by Chiricahua Apaches, but the impacts probably were limited (Kaib et al. 1999). However, more than a century of settlement has altered the dynamics in these ecosystems, resulting in declines in native herbaceous species, less frequent fires, and increases in woody species. The loss of a continuous herbaceous cover

limited the spread of natural fires and allowed seedlings of woody species to become established. Prescribed fire is seen as a technique for restoring the natural processes within the savannas.

The U.S. Department of Agriculture (USDA) Forest Service's Coronado National Forest and its partners have just completed a draft for the Peloncillo Programmatic Fire Plan to address fire management strategies for the Peloncillo Mountain range, which lies along the southern Arizona–New Mexico border. One persistent question concerned the impacts of cool-season (November–April) and warm-season (May–October) prescribed fires on the oak ecosystems that are common throughout these mountains. Fires normally occur in June or early July before the summer monsoon rains. However, there are questions about the effects

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of different burning seasons and fire intensities on this ecosystem. Many managers would like to duplicate natural warm-season burning, but others prefer low-severity, cool-season burning in the early spring or winter. Severe warm-season fires could destroy vegetation and litter, result in soil water repellency, and impact important wildlife habitats. Hotter warm-season fires could damage important wildlife habitats by killing larger numbers of standing trees and shrubs used as nesting sites, thermal or hiding cover, or species such as Palmer agave (*Agave palmeri*) that provide nectar for endangered lesser long-nosed bats (*Leptonycteris curasoae*) (Slauson et al. 1999). The potential conversion of large areas of woodlands or savannas to open grasslands would affect the habitat of the threatened New Mexico ridge-nosed rattlesnake (*Crotalus willardi obscurus*), and fire potentially could cause snake mortality (Holycross et al. 1999). Cool-season fires, however, could leave the soil surface without a protective cover for a longer period, resulting in increased erosion because most native herbaceous species grow and germinate during the summer monsoon period.

The USDA Forest Service's Rocky Mountain Research Station's Southwestern Borderlands Ecosystem Management Project and its cooperators have initiated a research program to evaluate the impact of season of burning using an ecosystem approach in which a large number of ecosystem components, including hydrology, sedimentation, vegetation, soil nutrient dynamics, small and large mammals, birds, and snakes and other reptiles are evaluated. An ecosystem approach to land management requires an evaluation of the blend of physical and biological factors needed to assure productive, healthy ecosystems (Kaufmann et al. 1994). Ideally, social and economic values also should be considered.

Encinal or oak woodlands and savannas together cover >80,000 km² in the United States and Mexico. Much is known about the oak woodlands but less about oak savannas. Ecological, hydrological, and environmental characterizations of the woodlands have been obtained through collaborative efforts involving a large number of people (McPherson 1992, 1997; DeBano et al. 1995; Ffolliott 1999, 2002; McClaran and McPherson 1999). Comparable characterizations of the lower-elevation oak savannas and open woodlands are needed to enhance the knowledge of all of the oak ecosystems in the Madrean Archipelago region. Oak savannas differ from oak woodlands in that their structure is more open with fewer trees and, as a consequence, a higher level of herbaceous production might be expected. We describe studies and preliminary results obtained during the pre-treatment phase of the project and provide information on hydrologic and erosion-sedimentation characteristics, tree overstory species composition and density, herbaceous understorey production, and observations of indigenous wildlife species.

STUDY AREA

Twelve small watersheds were selected in the oak savannas on the eastern side of the Peloncillo Moun-

tains of southwestern New Mexico to evaluate the impacts of cool- and warm-season prescribed burning treatments on the hydrological and ecological characteristics of the watersheds (Gottfried et al. 2000) and collectively form the study area (Figure 1). These watersheds, called the Cascabel watersheds, cover a total area of 182.6 ha. Individual watersheds range in area from approximately 24.0 ha (Watershed E) to 7.6 ha (Watershed G). The area is typical of oak savannas and open woodlands of the region. The Cascabel Watershed study area (lat 31°33'N, long 108°59'W) is largely within the Douglas Ranger District of the Coronado National Forest and adjacent to the Diamond A Ranch in New Mexico. The watersheds are situated between 1,640 and 1,705 m in elevation, and the nearest long-term precipitation station indicates that annual precipitation averages 597 mm, with nearly one-half occurring during the summer. Geological, physiological, and vegetation characteristics of the Cascabel watersheds have been previously described by Gottfried et al. (2000, 2002) and Neary and Gottfried (2004).

EXPERIMENTAL DESIGN AND SAMPLING METHODS

Physical Components

Hydrology

The six watersheds on each side of the ridge were divided into two groups of three (Figure 1). Each set will have the two burning treatments and an untreated control treatment that will be assigned randomly. The effects of treatments will be assessed by comparing the changes between pre- and post-treatment observations on the treated watersheds to comparable observations on the untreated watersheds. Streamflow data from individual storms will be the primary parameter in the analysis of treatment effects. The watersheds currently are being calibrated and treatments will be initiated once sufficient data are collected to describe pre-treatment conditions. The current 6-y drought has prolonged the calibration period.

Each watershed contains two Parshall flumes: a small flume with the capacity of 122 L/s to measure common low flows and a larger flume to measure anticipated large flows. Eight of the larger flumes have a capacity of 1,628 L/s and four have a capacity of 1,209 L/s (Gottfried et al. 2000). A sediment dam was constructed in each watershed, and permanent channel cross-section stations have been established and measured. Two complete weather stations and six supplemental recording precipitation gauges have been established throughout the watershed area.

Erosion and Sedimentation—Channel Characteristics

An early decision was to determine the character of all the channels in the 12 Cascabel watersheds because of the observed differences in channel conditions and the wide range of values in peak discharges that were estimated by using field measurements and U.S. Geological Survey and Natural Resources Con-

Cascabel Watershed Boundaries and Treatment Types

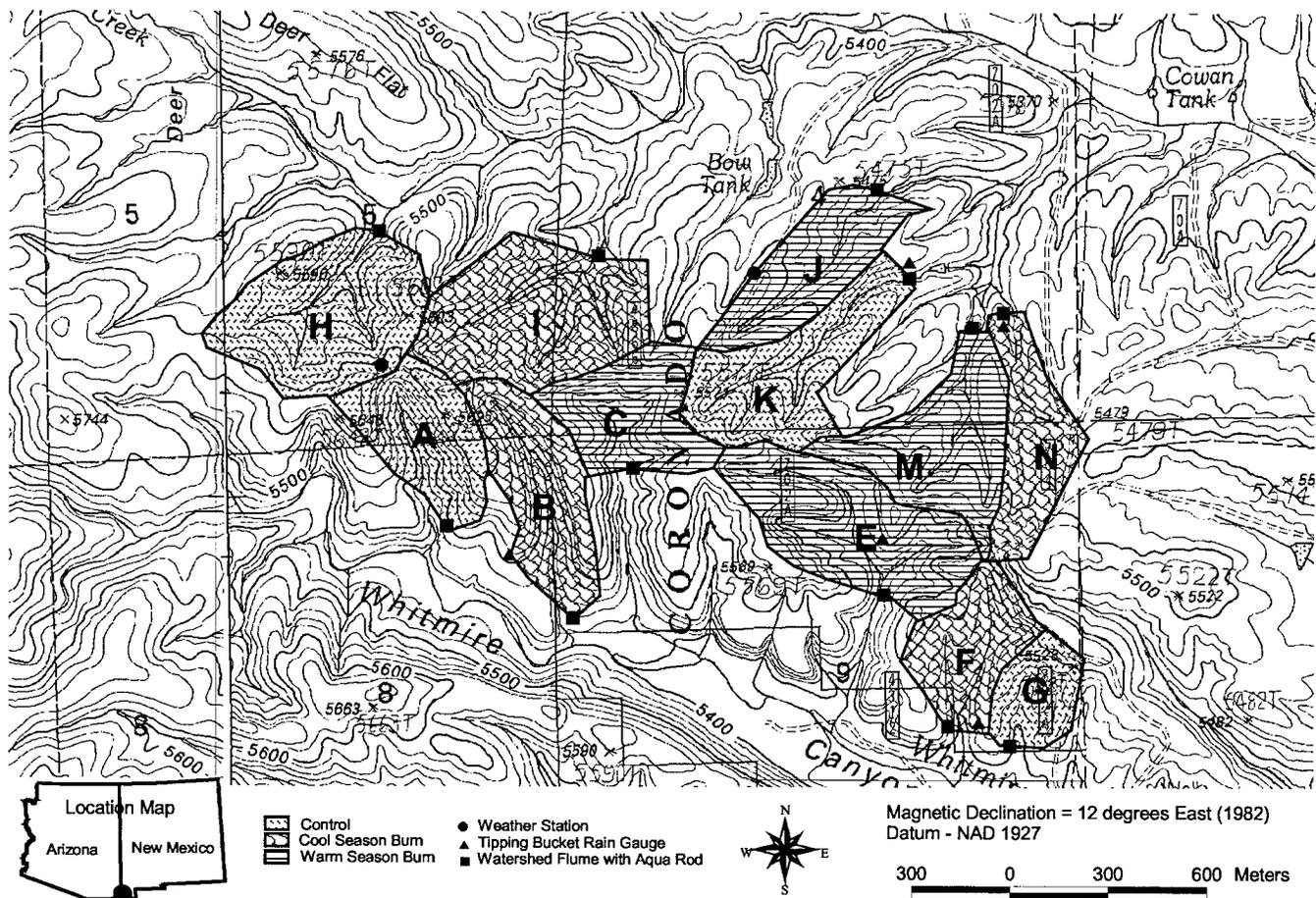


Fig. 1. The 12 Cascabel watersheds, Peloncillo Mountains, southwestern New Mexico, were divided into four groups of three watersheds and the potential burning season and control treatments were randomly assigned. Watersheds A, G, H, and K have been designated as controls, watersheds B, F, I, and N will receive warm-season burns, and watersheds C, E, J, and M will receive cool-season burns. Another analysis following the pre-treatment period may require that some treatments be reassigned among watersheds.

servation Service methods (Gottfried et al. 2000). A consensus opinion was reached among the principal investigators that the physical condition of the channels might strongly influence hydrologic responses, especially where long reaches of channel consisted mainly of bedrock outcrops.

Line-transect surveys were conducted on all channels, including side channels, starting at the larger Parshall flume and continuing upslope along the channel until the channel was no longer discernable. Lengths of channel were classified as bedrock, coarse alluvium, fine alluvium (sand), vegetation, woody debris, and other. Accumulated lengths of each class were summed for each watershed.

Side-Slope Erosion

Three erosion pins were installed in May 2004 around every third sample point used for vegetation and wildlife studies on each of the watersheds. These formed the basis to estimate soil erosion. Two erosion pins were placed about 1.8 m upslope and one erosion pin was placed 1.8 m downslope of the points. A total

of 438 erosion pins was installed on the study area. Initial measurements of soil loss were made in October 2004 following the summer monsoon rains of the year. The erosion pins were reset after this measurement to facilitate estimates of future soil loss. A bulk density value obtained from soil samples collected on the watersheds was used as the basis to convert measurements of average soil loss on a watershed to corresponding erosion rates in terms of metric tons per hectare.

Biological Ecosystem Components

Vegetation

On each of the 12 Cascabel watersheds, between 35 and 45 sample points were located along transects perpendicular to the main stream system and extended from ridge to ridge. Intervals between the sample plots varied among the watersheds depending on the size and configuration of the watershed sampled. A total of 421 sample points were located on the watersheds. Measurements of tree overstory conditions and esti-

mates of herbaceous production were obtained on plots centered over these sample points. Species composition and density of the tree overstory were measured on 0.10-ha circular plots. Single-stemmed trees were measured in terms of their diameter root collar (DRC) and multiple-stemmed trees in equivalent diameter root collar, following procedures outlined by Chojnacky (1988). Tree density is expressed in stems per hectare. Dead and down woody material was measured at the points using the survey technique developed by Brown (1974). Production (standing biomass) of grasses, forbs, and shrubs in the understory was estimated using 0.89-m² plots. These components of herbage production were estimated by weight-estimate procedures (Pechanec and Pickford 1937) in the spring (early growers) and fall (late growers).

Wildlife

Accumulations of fecal droppings (pellets) were tallied and then cleared on 0.004-ha plots centered over sample points in the spring (late May) and fall (early October) to assess the occurrence of white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*), and desert cottontail (*Sylvilagus audubonii*). Bird sightings in a 5-min observation period were tallied by species at the sample points to characterize avifauna on the study area during a week in late May and a week in early October.

RESULTS AND DISCUSSION

Physical Components

Hydrology

Hydrological measurements were started in 2001; unfortunately, the continuing regional drought has limited the number of streamflow events during the past 4 y. A total of 21 events have been measured but only two, occurring in January 2005, were measured on all 12 watersheds. The January storms, which produced about 73.7 mm of precipitation, produced peaks on all watersheds, with a range of 45.3 L/s on Watershed H to 184.0 L/s on Watershed I. Peak flows of between 96.3 and 150.1 L/s were recorded on several watersheds during a summer rainstorm on 31 August 2004. Additional peak flow events from the summer of 2005 currently are being evaluated.

Channel Characteristics

Channel distances ranged from 420 m for Watershed A (Figure 1) to 1,763 m for Watershed E, which has two main forks; the mean distance for the 12 watersheds is 1,020 m (Neary and Gottfried 2004). There appears to be a great disparity in percentage of individual watershed channel characteristics (Figure 2), although differences have not been evaluated statistically. Rock-lined areas occur within 6.4% of the channel in Watershed A and 41.0% in Watershed K. There was also a large range between all the Cascabel watersheds in fine alluvium (0.2–35.7%) and coarse alluvium

(18.3–74.0%). Coarse alluvium channel material is the most common characteristic dominating all watershed channels except watersheds K and N. The type of channel substrate could make a significant difference in total water yields as well as storm peak flow responses. The watersheds with the higher percentages of bedrock channels are apt to be “more flashy” in nature, producing peak flows more rapidly than watersheds with less rock. The watersheds with the higher percentages of alluvium in their channels might prove to be less flashy and have lower but sustained flows because their channels would have a larger in-channel water storage capacity. There was far less variation among watersheds in the remaining three categories (vegetation, woody debris, and other), which have been combined in Figure 2. Channels on watersheds F and G had the greatest amounts of vegetation and woody debris.

Side-Slope Erosion

The first known estimates of surface erosion on a watershed basis in the oak savannas of the southwestern United States have been obtained on the Cascabel watersheds. Two measurements of soil erosion are currently available, one following the summer monsoon season of 2004 and the second in the spring of 2005 after the winter precipitation period. Estimated soil erosion averaged 23.8 t/ha on the 12 Cascabel watersheds (collectively) following the summer monsoons, with estimates on individual watersheds ranging from 10.8 to 38.3 t/ha. Soil erosion after the winter precipitation period, which included several high-intensity precipitation events, was estimated to be 27.1 t/ha, ranging from a low of 7.8 to a high of 40.6 t/ha. For the year of measurement, therefore, the soil erosion rate was 50.9 t/ha. No meaningful relationships between the magnitude of the estimated soil erosion on the individual Cascabel watersheds and the corresponding watershed size, stream-channel network, and physiography (slope position, slope percent, aspect) were evident in this initial analysis. A longer period of measurement is needed to detect these relationships if they exist.

Biological Ecosystem Components

Tree Overstories

The dominant species tallied on the Cascabel watersheds were Emory (*Quercus emoryi*) (60.1% of all trees tallied), Arizona white (*Q. arizonica*; 11.9%), and Toumey (*Q. toumeyi*; 4.4%) oaks and alligator juniper (*Juniperus deppeana*; 15.3%). Minor components are redberry juniper (*J. coahuilensis*; 2.0%), pinyon (*Pinus discolor*; 5.6%), and mesquite (*Prosopis glandulosa*; 0.7%).

Tree densities on the study area were summarized by total number and volume of all trees in size-class categories listed by O'Brien (2002) for the woodland types of the southwestern United States: saplings (2.5–12.4 cm DRC), medium trees (12.7–22.6 cm DRC), and large trees (22.9 cm DRC and larger). There were

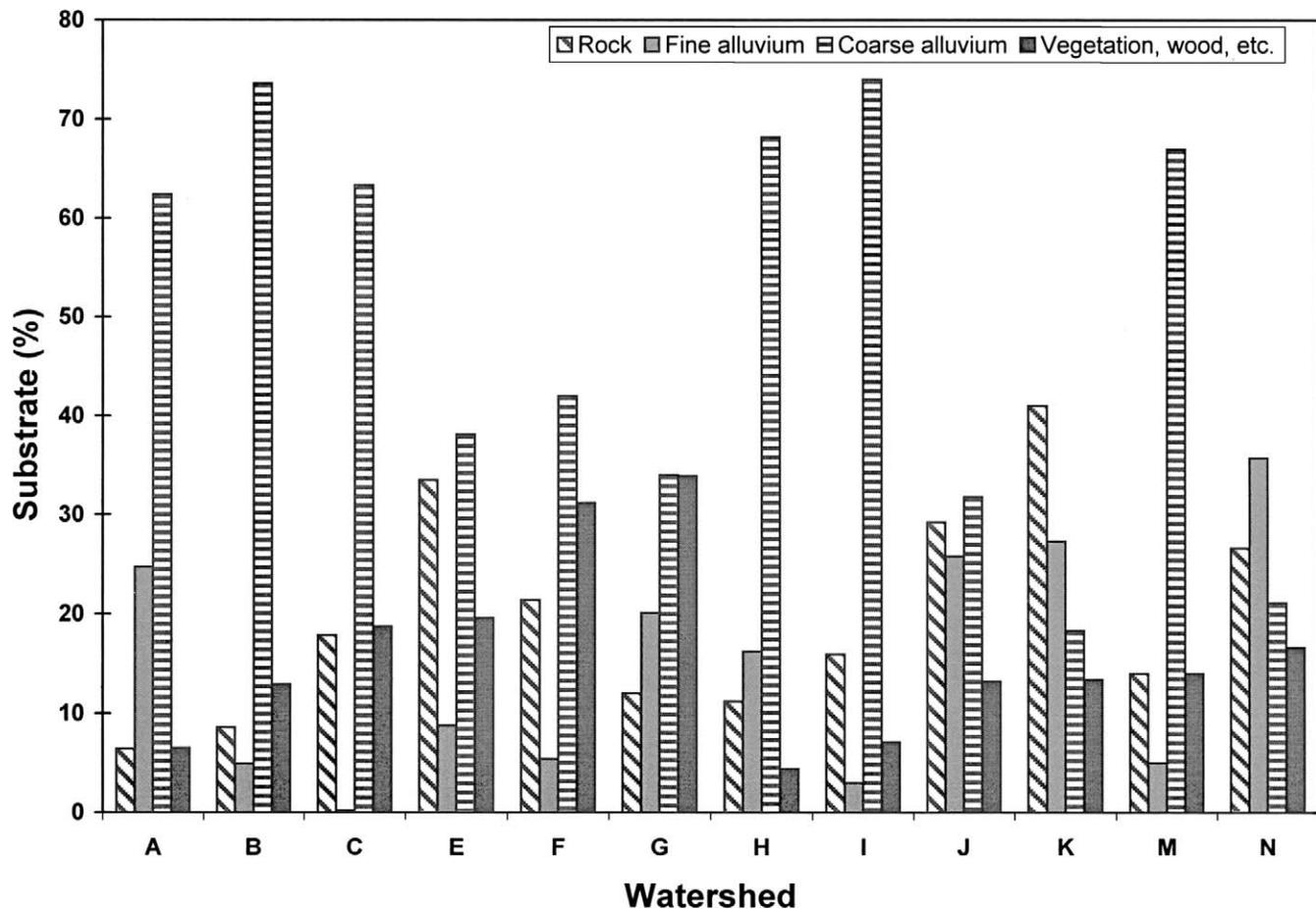


Fig. 2. Channel substrate characteristics of the 12 Cascabel watersheds, Peloncillo Mountains, southwestern New Mexico. These data are adapted from Neary and Gottfried (2004).

130.0 trees and 0.10 m³ of volume per hectare in saplings, 45.5 trees and 0.89 m³ of volume per hectare in medium trees, and 47.2 trees and 10.17 m³ of volume per hectare in large trees. Total densities were 222.7 trees and 11.16 m³ of volume per hectare. Approximately 58% of the trees were in the sapling class and the rest of the trees were almost equally divided between the two larger classes. These densities were generally less than observed in the higher-elevation oak woodlands that receive higher amounts of annual precipitation and support more than twice as many trees per hectare (Ffolliott and Gottfried 2005).

Dead and Down Woody Fuels

The average dead and down woody fuels on all watersheds combined was 1.61 t/ha. Nearly 90% of the material was Emory oak twigs and small branches. Most of the material was <1.2 cm in diameter and <0.6 m in length.

Herbaceous Understories

Included among the more commonly encountered grasses on the Cascabel watersheds were blue (*Bouteloua gracilis*), sideoats (*B. curtipendula*), slender (*B. repens*), and hairy (*B. hirsuta*) grama, bullgrass (*Muhl-*

enbergia emersleyi), common wolfstail (*Lycurus phleoides*), and Texas bluestem (*Schizachyrium cirratum*). Forb species, including species of verbena (*Verbena* spp.) and mariposa lily (*Calochortus* spp.), are comparatively minor herbaceous components. Buckbrush (*Ceanothus* spp.), beargrass or sacahuista (*Nolina microcarpa*), fairyduster (*Calliandra eriophylla*), and common sotol (*Dasylyrion wheeleri*) are scattered shrubs and half-shrubs on the area.

Total production of grasses, forbs, and shrubs on the Cascabel watersheds in the spring and fall of 2003 was 260.8 and 218.3 kg/ha, respectively. The corresponding levels of herbage production in 2004 were 198.5 and 202.3 kg/ha, respectively. We assumed that observed herbage production estimates reflect the influence of prolonged drought in the region. Estimates of herbage production continue to be made to adequately reflect variability in precipitation amounts and distribution on the study area.

The predominantly oak overstories in the savannas on the watersheds do not appear to affect herbage production. This observation is contrary to the commonly reported relationship of increasing herbaceous cover with decreasing overstory density (Ffolliott and Clary 1982). The lack of similar overstory–understory relationships has been reported for other southwestern oak

ecosystems (Ffolliott and Gottfried 2005). However, the reverse situation might not be true. McClaran and McPherson (1999) reported that a dense cover of perennial grass can limit successful Emory oak regeneration on ungrazed sites in similar oak ecosystems. Further study is necessary to verify the overstory–understory relationships or to identify other important variables, such as crown characteristics, that may influence herbaceous production.

Utilization of the herbaceous plants by herbivores has averaged <5% in both years of study. However, cattle had been removed from the Cascabel watersheds and the surrounding areas by local ranchers for several months in the summer and early fall of 2004 because of the prolonged period of below-average annual precipitation. Populations of indigenous herbivores have also been low on the watersheds in recent years.

Wildlife

Abundance and habitat preferences of birds and some of the other wildlife found in the densely structured oak woodlands of the southwestern United States are known for some of the representative species, but little information on these topics is available for the more open oak savannas of the region. White-tailed deer on the Cascabel watersheds are largely scattered throughout the year. However, population estimates are unreliable because of the limited time that pellet counts have been tallied. Patterns of repeated observations of pellet accumulations and habitat conditions (vegetation, physiography, etc.) at the sample points were inconclusive. Findings and conclusions for desert cottontail were similar. A larger number of birds and a greater diversity of species were observed on the Cascabel watersheds in the spring (May) than in the fall (October) (Jones et al. 2005). While some species, such as bushtit (*Psaltriparus minimus*), Mexican jay (*Aphelocoma ultramarina*), juniper titmouse (*Baeolophus ridgwayi*), mourning dove (*Zenaida macroura*), and scaled quail (*Callipepla squamata*), were tallied in both observation periods, other species such as dusky-capped flycatcher (*Myiarchus tuberculifer*), northern mockingbird (*Mimus polyglottos*), and turkey vulture (*Cathartes aura*) were observed in spring but not in fall. Montezuma quail (*Cyrtonyx montezumae*) were observed only in the fall. It remains to be seen if this pattern of abundance and species diversity continues to persist on the watersheds in the future. No meaningful relationships between bird sightings and the habitat conditions represented by the sample points were observed.

CONCLUSIONS

Researchers and land managers are attempting to learn more about the oak ecosystems of the southwestern United States and northern Mexico so that sound land stewardship can be maintained or enhanced. Learning of the impacts of the planned reintroductions of fire into these ecosystems is a primary component in these efforts. Studies on the hydrology

and erosion–sedimentation dynamics and the related biological impacts of prescribed fire treatments in the oak savannas in the southwestern Borderlands is a focus of this research project. We described initial results of the pre-treatment phase of the study. Experimental designs and sampling methods were selected to be able to detect statistical differences in the measured ecosystem characteristics between the pre-treatment and post-treatment conditions. The calibration period will be monitored until a sufficient number of storm events is observed to provide estimated statistical power of 0.90 to detect hydrologic changes of 100% with a significance of 0.05. Analyses of the biological resources responses assume a level of significance between 0.15 and 0.20.

Detectable changes can only be surmised at this time. Nevertheless, we anticipate post-treatment changes in the timing of streamflow and discharge rates. Side-slope erosion rates are likely to increase, leading to changes in channel conditions. Species compositions might change and the production of grasses, forbs, and shrubs will likely increase after the prescribed fire treatments as a result of the decrease in tree overstory densities and removal of litter and duff layers. Increased browse should be available to deer populations because the oak species on the watersheds readily sprout following fire. Ground-nesting avifauna is anticipated to decrease in numbers and diversity, while avifauna nesting in standing dead trees (snags) should increase. Reducing the loadings of dead and down woody material and other fuel fractions, one of the main objectives of the prescribed fire treatments, should occur. All of these possible changes will be evaluated and contrasted within the context of the planned cool-season and warm-season fires.

Detection of these changes, if they occur, will be determined in two ways. If post-treatment weather regimes are similar to those encountered in the pre-treatment phase—the latter representing a drought condition—comparisons of before and after ecosystem characteristics should provide the necessary information. However, if the two weather regimes are significantly different, comparison will be made between changes on burned and unburned control watersheds. Both approaches are planned in an effort to provide the most comprehensive picture of the effects of prescribed fire in the oak savannas.

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VEGETATION RESPONSES TO SEEDING AND FERTILIZATION TREATMENTS AFTER WILDFIRE IN NORTH-CENTRAL WASHINGTON STATE

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ABSTRACT

As part of Burned Area Emergency Response, the USDA Forest Service and other federal land management agencies often prescribe slope stabilization treatments for steep, severely burned slopes in forests and rangelands to reduce risks of erosion. Despite widespread acceptance and use of these slope stabilization treatments, their efficacy for increasing vegetation cover and reducing soil erosion has not been well established.

In fall 2002, we established a field experiment on severely burned areas of the Deer Point Fire in north-central Washington State to examine the effects of two slope stabilization treatments, seeding and fertilization, on development of vegetative cover and recovery of native vegetation during the first 3 y after fire. We were also interested in longer-term effects on species diversity and the recovery of native plant communities. At each of four sites, we established 32 or 64 experimental plots (48 m²) and assigned plots to one of four seeding treatments (winter wheat, mixed perennial grasses, both, or neither) and one of two fertilization treatments (fertilization or nothing) in factorial combination. We measured plant functional group cover during each of the first three growing seasons after fire, and plant cover by species during the first and third growing seasons after fire.

Sprouting shrubs and grasses provided a large proportion of total vegetation cover during the first year post-fire. The winter wheat seeding treatment was generally ineffective and had no significant effect on vegetative cover. The mixed-perennials seeding treatment increased mean herbaceous plant cover and total vegetation cover by 4–5% at three of four sites during the first growing season after fire; however, differences in total vegetation cover were statistically significant at only one site. Fertilization treatments altered the relative cover of herbaceous and woody plants, but produced a significant increase in total vegetation cover at only one site. Seeding treatments appeared to be most necessary and effective on sites with sparse pre-fire understory vegetation and sparse soil seed banks. More experimentation and better monitoring is needed, however, to select appropriate species (preferably native) for different habitats.

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MANAGEMENT CHALLENGES AND OPPORTUNITIES RELATED TO PRAIRIE GROUSE; A PICTORIAL OVERVIEW

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ABSTRACT

Grouse species occupying the Great Plains and associated habitats include greater sage-grouse (*Centrocercus urophasianus*), Gunnison sage-grouse (*C. minimus*), greater prairie-chicken (*Tympanuchus cupido pinnatus*), lesser prairie-chicken (*T. pallidicinctus*), and sharp-tailed grouse (*T. phasianellus*). All of these species have experienced significant declines in numbers during the last half century. The heath hen (*T. cupido cupido*) and Attwater's prairie-chicken (*T. cupido attwateri*) are extinct and almost extinct, respectively.

Management challenges and opportunities associated with plains grouse declines and potential recoveries are myriad and include but are not limited to various facets of livestock management, farming activities, farm conservation programs, use of fire, habitat fragmentation, predation, harvest, road management, power lines, towers, fences, resource extraction to include wind power, forestry practices, habitat conversion, mowing, chemical use, noxious weeds, hydrology, development, recreation, habitat acquisition, and easements.

The North American Grouse Management Plan considers these challenges, and this presentation provides pictorial understanding of many of the problems involved as well as a brief discussion of associated impacts. Rather than an *Exxon Valdez* or Mount St. Helens type of catastrophic destruction, habitat in the Great Plains is being slowly chipped away, piece by piece, with little planning or regulation. In many cases, multiple factors combine as threats to synergistically impact grouse survival. There are many fire-related categories such as the brush-control problems in South Texas where we are trying to reintroduce Attwater's prairie-chicken. Fire could solve this problem, but there is great reluctance to use it. A similar situation exists with sharp-tailed grouse where forest-succession problems in the north country can completely eliminate once suitable sharp-tailed grouse habitat unless fire or mechanical tree removal is utilized. Again, there is a reluctance to use fire due to perceived danger. Fire is only one aspect in the revival of grasslands, but an important one. As with numerous other management concerns—opportunities, fire is essential for management that results in healthy habitats. Yet too much or too little, both spatially and temporally, can result in a decidedly negative impact for management of grouse of the plains.

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LEKING AND NESTING RESPONSE OF THE GREATER PRAIRIE-CHICKEN TO BURNING OF TALLGRASS PRAIRIE

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ABSTRACT

In the southern Great Plains, the greater prairie-chicken (*Tympanuchus cupido*) is confined to tallgrass prairie, a habitat now largely converted to agriculture. Remaining prairie is highly fragmented and subjected to land management practices that greatly alter the ecosystem of this species. Chief among these practices are deliberate, large-scale spring burns associated with early intensive stocking of cattle. We used extensive data to infer how such fires affect the prairie-chicken's lekking and nesting behavior. From 1998 to 2000, 60–79.4% of our study area—a 45,000-ha expanse of tallgrass prairie in north-central Oklahoma—was burned in spring. Prairie-chickens tended to lek on unburned areas but not in a pattern that differed from random habitat choice. Leks on burns tended to be <200 m from unburned prairie. Females strongly avoided nesting in areas burned in spring 1998 and 1999 ($n > 25$ nests/y). Nesting effort was poor in 2000, prohibiting statistical analysis, but 5 of 8 nests were on unburned prairie. Incorporating data from 1997 ($n = 12$ nests), only 14 of 74 nests were placed on burned prairie, and only 5 of the 64 nests from 1998 to 2000 were located on prairie burned all three of those years. Avoidance of burns was particularly strong before June. Despite strong avoidance, nest success did not differ between burned or unburned prairie. Our findings raise two concerns: 1) if leks are established only <200 m from unburned prairie, then an increase in the proportion of burns may inhibit lek formation, and, more important, 2) if females avoid nesting on recent burns, then an increase in the proportion of burns may cause a female to concentrate nesting effort on small patches (or forego nesting). We recommend moderation of spring burning (e.g., patch burning) of tallgrass prairie, lest the greater prairie-chicken be driven further from its already piecemeal habitat.

keywords: greater prairie-chicken, leks, nest placement, Oklahoma, spring burns, tallgrass prairie, *Tympanuchus cupido*.

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INTRODUCTION

Only about 4% of the original extent of tallgrass prairie remains (Samson and Knopf 1996). Birds and other organisms dependent on this habitat have declined concomitantly, and many species have become of high conservation concern. Emblematic of this habitat is the greater prairie-chicken (*Tympanuchus cupido*), a species whose fate has followed the prairie's fate. Even where tallgrass prairie remains—where it has not succumbed to the plow—it is impacted by land management, particularly those practices associated with cattle ranching (Robbins et al. 2002). Chief among these management tools is prescribed fire.

Fire is an integral component of the prairie ecosystem (Collins and Wallace 1990), with natural fires occurring in tallgrass prairie once or twice per decade (Reichman 1987). Currently, most natural fires are suppressed, yet prescribed fires are used to produce more forage for cattle, particularly as a means of implementing early intensive stocking (Smith and Owensby 1978). Spring burning generates greater plant growth (Hadley and Kieckhefer 1963, Hulbert 1988),

in part by removing rank vegetation that limits productivity (Knapp and Seastedt 1986), but may offer cover for organisms inhabiting the grassland. Relative to unburned areas, recently burned areas often harbor significantly greater abundance or biomass of herbivorous arthropods, particularly grasshoppers (Orthoptera: Acrididae) (Warren et al. 1987, Evans 1988, Swengel 2001, Shochat et al. 2005).

Wright (1974:8) asserted that the “prairie chicken . . . [is] favored by fires which create variety in habitat,” but whether this variety is beneficial to the species is an open question. Moreover, there may be tradeoffs associated with burning; for example, increases in arthropod biomass may benefit the greater prairie-chicken, but a loss of cover may hurt it, a similar tandem of benefit and cost described for the closely related lesser prairie-chicken (*T. pallidicinctus*; Boyd and Bidwell 2001). In addition to tradeoffs, different sexes or life stages may respond differently to burned habitat.

Using extensive lek and nest data from a 4-y study of habitat use and reproductive ecology of the greater prairie-chicken, we determined whether this species responds positively, negatively, or not at all to burning of the tallgrass prairie in spring. Although availability

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of lek habitat is rarely limiting, it is possible that males avoid establishing leks on recent burns; thus, we determined if lek placement (and lek size) was associated with burning. Likewise, nesting females may avoid burns, so we examined whether nest placement was associated with burning. We further examined how nest success, clutch size, brood size, and brood placement associated with burns.

STUDY AREA

The study area encompassed approximately 45,000 ha of tallgrass prairie in north-central Osage County, Oklahoma, its northern edge abutting Kansas (lat 36°46'–37°00'N, long 96°22'–96°40'W). The area was in a southern extremity of the Flint Hills geological system, which extends through central Kansas southward from near the Nebraska border. The Flint Hills comprise largely unplowed (soils are underlaid with rock) tallgrass prairie, though much of this region is grazed heavily and burned annually (Zimmerman 1997, Robbins et al. 2002). Habitat in our study area was relatively homogenous prairie, with no cultivation, no significant development, and few fences. The few roads were primarily graded dirt or gravel without shoulders. Deciduous woodland (<5% of the area) occurred only in a small portion of the southeastern corner of the area and in narrow corridors along two creeks.

Habitat on the study area was tallgrass prairie. Prescribed fire created some spatial and temporal heterogeneity in the study area. Cattle grazing usually followed burning (see below), the typical grazing system being early intensive stocking: steers are brought to the ranches for approximately 100 d from April to July, allowing the range to recover in autumn and winter (Smith and Owensby 1978). Cow-calf operations occupied approximately 10% of the study area; such operations avoid annual burns and graze at a lower stocking rate but continue throughout the year. A low density of American bison (*Bison bison*) grazed approximately 5% of the study area year-round, all of it on The Nature Conservancy's Tallgrass Prairie Preserve (the southeastern quarter of the study area). Patches (<100 ha) of this preserve were burned sporadically. A small fraction (<1%) of the study area was hayed each year. Rainfall during our study was higher (112.7–134.7%) than the long-term (1949–2003) average ($\bar{x} \pm SD = 97.0 \pm 23.6$ cm/y), but annual temperature was average (97.1–104.0%).

Between 60% and 80% of the study area was burned annually, typically in early spring (March–April). We mapped the extent of these burns in 1998, 1999, and 2000, each year between 1 January and 31 May. We classified an area as “burned” only if it had burned since the previous growing season (i.e., burns >1 y old were excluded). Ground maps were transferred into ArcView 3.3 (ESRI Software, Redlands, CA) using base maps from Rea and Becker (1997).

METHODS

Radio Tracking

We tracked radio-tagged greater prairie-chickens year-round from April 1997 to July 2000. Birds were trapped on leks using walk-in funnel traps connected by 8 m of plastic drift fence arrayed in a zigzag pattern. Each bird was fitted with a bib-mounted radio-transmitter and a loop antenna (AVM and Telemetry Solutions [ATS], Concord, CA) weighing 18 g, approximately 2% of a prairie-chicken's body mass (800–1,000 g). Tracking equipment consisted of 5-element, handheld Yagi antennas and ATS model R2000 or R4000 receivers. On average we tracked a bird once every 3 d at varying times of day. For all-day tracking, conducted sporadically throughout the study, we recorded a bird's location at least once every 30 min, more frequently if a bird moved >1 km. The vast majority of bird locations were from direct homing; <1% of locations were from triangulation. Two person-days per week were devoted to finding “lost” birds—i.e., individuals not detected for 2 weeks. We also conducted wide aerial transects for lost birds 5–6 times/y, extending 2–3 km past known bird locations.

We surveyed the study area for active leks from 27 March to 7 May each spring, always between 30 min before dawn and 2.5 h after dawn and only on mornings with good weather. We spent an hour on each section surveyed and recorded the number of birds at each lek.

If a female tracked in spring occupied the same location for 2 consecutive days or was otherwise thought to be nesting, we approached her cautiously to determine if she was on a nest. If so, we placed a marker radio at the nest's location so the nest could be monitored from afar, thus minimizing disturbance to the female. We monitored all nests every 2–3 d. We did not routinely flush a female to determine nest contents, but if a female flushed of her own accord or if she had been incubating for several weeks, we gathered data on clutch size (66 of 74 nests).

Statistical Analyses

We examined lek and nest placement under a null model assumption that if either was random then each would occur in the same proportions as those of burned versus unburned prairie. This model assumes that all of the study area was available for use, an assumption we think is valid given that the majority of the study area is tallgrass prairie with little soil disturbance or woody vegetation (see Study Area). Under this null model, lek or nest placement could be tested with a χ^2 goodness-of-fit test. All statistical tests can be found in Sokal and Rohlf (1995).

We assessed site fidelity—here defined as the tendency for leks to have the same geographic center across years—using Cochran's Q , a nonparametric test for repeated-measures data that are dichotomous (i.e., a lek was present or absent from a particular location in a given year). We used the nonparametric Wilcoxon

rank-sum test to determine if lek size tended to differ on burned versus unburned prairie.

For the timing of nest placement, we assessed nesting before and after 1 June using a Mantel–Haenszel χ^2 , a test of frequencies divided into discrete groups (in this case time of year). We assessed differences in nest date with a one-way analysis of variance (ANOVA) with date as the response and burned versus unburned as the categorical variable. For ANOVAs, we report effect size as *d* (Cohen 1988).

Our assessment of reproductive effort was several-fold. We constructed survival curves for nests using a Kaplan–Meier product-limit estimator and compared curves on burned versus unburned prairie with a log-rank χ^2 test. We compared productivity by burn treatment for clutch size and fledgling production. In both cases we treated nest date as a covariate and compared the response variables with an analysis of covariance (ANCOVA).

RESULTS

Lekking Behavior

The geographic center of leks tended to shift from year to year, such that birds did not return to fixed locations but moved according to changing conditions. As a result, site fidelity was not a strong factor in lek location, in that interannual locations were only marginally correlated (Cochran's *Q*-test: $Q = 4.33$, $df = 2$, $P = 0.11$), a finding consistent with past studies (e.g., Merrill et al. 1999). Male prairie-chickens tended to lek on unburned prairie (Figure 1), but they did not choose unburned habitat differently from a random choice of available unburned habitat (goodness-of-fit test: $0.22 < \chi^2 < 4.46$, $df = 1$, $P > 0.10$, $20 < n < 27$ leks). Even so, leks on burned prairie tended to be < 200 m from unburned prairie: 54 of 79 leks (68.4%) surveyed during 1998–2000 had unburned prairie somewhere within this radius, including on the lek itself. Only 21 of 55 (38.2%) leks were on such land burned in each of the three springs from 1998 to 2000. Regardless, lek size tended to be larger on burned ($\bar{x} \pm SD = 7.4 \pm 5.6$ birds) than on unburned ($\bar{x} \pm SD = 5.1 \pm 3.7$ birds) prairie (Wilcoxon rank-sum test: $W_s = 1,216$, $df = 1$, $P < 0.06$).

Nesting Behavior

Females strongly avoided nesting in areas burned in spring 1998 (Figure 2A; $\chi^2 = 64.84$, $df = 1$, $P < 0.0001$, $n = 28$ nests) and in spring 1999 (Figure 2B; $\chi^2 = 29.64$, $df = 1$, $P < 0.0001$, $n = 26$ nests). Nesting effort was poor in 2000, prohibiting statistical analysis, although 5 of 8 nests were on unburned prairie even though 63.6% of the area had been burned that spring. Incorporating data from 1997 ($n = 12$ nests), only 18.9% (14 of 74) of the nests were placed on burned prairie, and only 5 of the 64 (7.8%) nests from 1998 to 2000 were located on prairie burned in each of the three years.

Females particularly favored nesting on unburned

prairie before June (Figure 3; Mantel–Haenszel $\chi^2 = 17.81$, $df = 1$, $P < 0.0001$, $n = 74$), just after the approximate midpoint (29–30 May for our data) of the prairie-chicken's nesting season. Indeed, on average, nests were found on burned prairie ($\bar{x} \approx 07$ June, $SD = 17$ d) > 3 weeks later than they were found on unburned prairie ($\bar{x} \approx 12$ May, $SD = 10$ d), a significant difference in nest timing (ANOVA: $F_{1,70} = 55.94$, $P < 0.0001$, $n = 74$, $d = 2.23$).

Despite avoidance of burned prairie and even though nests on unburned prairie survived, on average, slightly longer than nests on burned prairie ($\bar{x} \pm SE = 15.4 \pm 1.2$ d vs. 13.9 ± 2.3 d), nest survival as a whole did not vary relative to burning (Figure 4). However, nests on unburned prairie were more productive, with clutch size averaging nearly 2 eggs greater and fledglings averaging nearly 5 greater (Table 1).

Determining whether smaller clutch sizes were an effect of burning per se is confounded by the diminution of clutch size with later laying date (linear regression: $F_{1,62} = 31.47$, $P < 0.0001$, $n = 64$, $r^2 = 0.34$) and renesting attempts (ANOVA: $F_{1,64} = 7.16$, $P < 0.01$, $n = 65$, $d = 0.96$), itself a function of laying date. Accounting for the effects of nest date—i.e., incorporating nest date as a covariate in a linear model—implies that neither burns (ANCOVA: $F_{1,62} = 0.26$, $P > 0.50$, $n = 64$) nor nest attempts (ANCOVA: $F_{1,62} = 0.02$, $P > 0.80$, $n = 64$) had an effect on clutch size. Likewise, once accounting for nest date, brood size was apparently not related to burning per se (ANCOVA: $F_{1,22} = 1.86$, $P > 0.10$, $n = 24$).

In contrast to nest survival, brood survival may be affected positively by proximity to a burn edge: 3 of 7 (43%) broods from nests > 500 m from a burn were depredated completely within 1 week of hatching, yet none of 13 nests < 500 m from a burn were lost completely.

DISCUSSION

Lekking Males

Our data suggest that lekking male greater prairie-chickens are not affected adversely by spring burns. Prairie-chickens lek in open, short habitat (Hamersstrom et al. 1957), so it is conceivable that regular burning (or grazing) is needed to create open patches with short grass or forbs that are favored for lek sites. This hypothesis is supported by our observation that larger leks tended to be on burned prairie. Even so, the tendency for leks on burned prairie to be near unburned prairie suggests that birds sought lekking locations within a short burst of flight from denser cover, presumably as a means of individuals reducing their risk of predation.

Regular burning may have an ancillary benefit. Greater prairie-chickens avoid lekking near forested habitat (Merrill et al. 1999), perhaps including intrusive woody plants such as the invasive eastern red cedar (*Juniperus virginiana*), a “weedy” species currently claiming large tracts of tallgrass and mixed-grass prairie (Briggs et al. 2002). Regular fires inhibit



Fig. 1. Greater prairie-chicken leks (dots) in (A) 1998, (B) 1999, and (C) 2000 in relation to spring burns, Osage County, Oklahoma. Dot size varies according to lek size (small dot = 1–5 males; medium dot = 6–10 males; large dot = 11–25 males). The heavy black outline delineates the study area. Unburned patches are shaded gray.

the spread of woody vegetation and promote the spread of grass and forb species, thus contributing to a healthy prairie ecosystem and its component avifauna (Reinking 2005).

Nesting Females

Nesting females strongly avoided prairie burned that spring (Figure 2), at least for their first nest attempts (Figure 3), but they preferred burned prairie for subsequent nest attempts; specifically, after May grass-

land vegetation had recovered from a burn to a sufficient extent (see Robel et al. 1998) that females were less apt to avoid burned areas. Although prairie-chicken nests may be lost to fire directly (Zimmerman 1997), we suggest that the major result of spring burns is an initial reduction in what females view as suitable nesting habitat. Many birds have facultative responses to conditions (Newton 1998), meaning they may nest “out of season” if conditions are favorable or may forego nesting “in season” if conditions are unfavorable (e.g., Bolger et al. 2005). We hypothesize that if

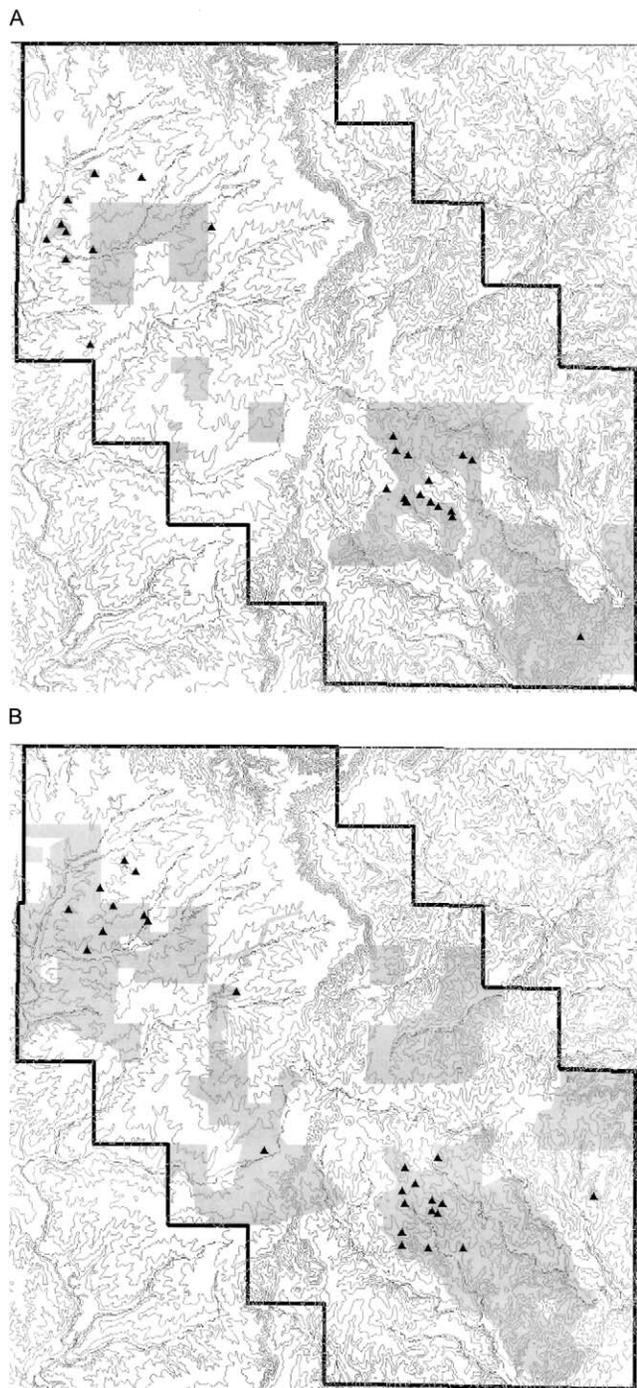


Fig. 2. Greater prairie-chicken nests (triangles) in (A) 1998 and (B) 1999 in relation to spring burns, Osage County, Oklahoma. The heavy black outline delineates the study area. Unburned patches are shaded gray.

prairie is burned extensively, females may wait to nest until they view conditions as being more favorable—in this case, until the prairie has recovered to an extent that a nest can be concealed adequately. Before fire suppression and wide cultivation of the prairie, large fires may have had a similar effect, but females then may have been able to disperse to suitable habitat contiguous with their home range. Extant prairie is cur-

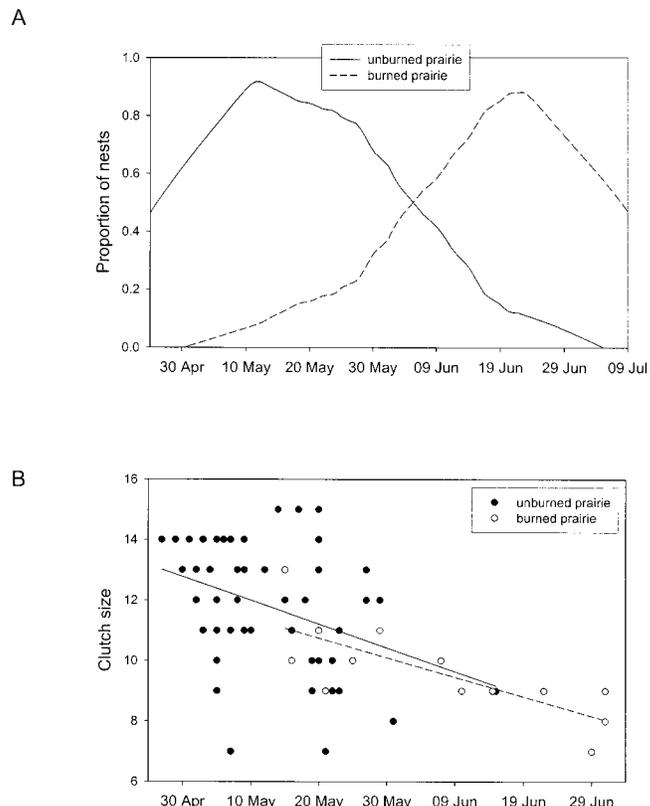


Fig. 3. Greater prairie-chicken nests, Osage County, Oklahoma, 1998–2000. (A) Proportion of nests in burned and unburned tallgrass prairie across the nesting season. Curves were derived from LOESS smooths ($f = 0.5$) of frequency data (accumulated nests per 5-d period). Vegetation recovers as the season progresses, so females are less likely to avoid burned areas later, with early June being a transition point. (B) Clutch sizes from individual nests plotted against nest date. Regardless of burning, regressions of clutch size against date have the same slope (solid = unburned, dashed = burned).

rently highly fragmented, and females may not be able to move readily to favorable unburned patches.

It is nearly universal that, within a species, avian clutch size decreases as the breeding season progresses (Klomp 1970, Winkler and Walters 1983). Our data show the same pattern for the greater prairie-chicken. Causes for this pattern are unclear, but if it is independent of effort—if clutch size is smaller later in the season regardless of whether the female is on her first, second, or third nesting attempt—then output will be lower by the time females begin to occupy areas burned that spring. Thus, there is reason to expect our observed pattern of lower reproductive output on burns.

We can only speculate why broods suffered apparently higher losses farther from burns. A plausible hypothesis is that accumulation of litter and tangled vegetation from the previous year's growth hampers brood movement, making them less able to escape a predator, an idea supported by our observation that most females that nested in unburned prairie typically moved their broods to recently burned prairie shortly after hatching. But movement to recent burns could be because female prairie-chickens preferentially select

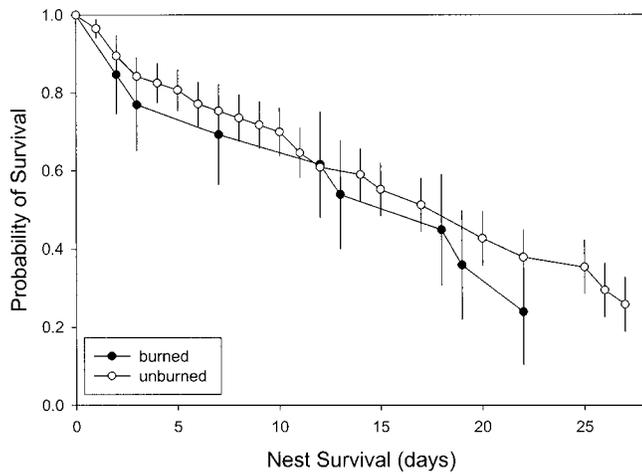


Fig. 4. Greater prairie-chicken nest survival in relation to spring burning of tallgrass prairie, Osage County, Oklahoma, 1998–2000. The Kaplan–Meier survival curve for nests on burned prairie appears to be lower, but standard errors overlap sufficiently to make the apparent difference insignificant (log-rank test: $\chi^2 = 0.29$, $df = 1$, $P > 0.50$, $n = 70$ nests).

prairie with higher levels of invertebrate biomass (Hagen et al. 2005), a feature of recent burns (Warren et al. 1987, Evans 1988, Swengel 2001).

If these hypotheses hold true and the observations remain consistent, together they present a conundrum for nesting greater prairie-chickens. Increased or continued extensive burning may cause females to forego nesting until a later date, by which time their reproductive potential will be diminished. Moreover, if broods require more open prairie to increase their odds of escaping predators, then females have another reason to wait for burns to recover or, at the least, to find suitably concealed nest sites (i.e., those in unburned prairie) near areas burned that spring. In either case, if burning is too extensive, then the predicted reduction in reproductive output on burned plots, a result of later nesting there, could reduce population size to a critically low level as a cumulative effect over time. Lastly, because our study was conducted during four years that, by chance, had higher than average rainfall, it is possible that burned prairie recovered more quickly than usual. We speculate that burns would recover later, and perhaps not to a sufficient extent, during dry years.

MANAGEMENT IMPLICATIONS

Our data suggest that both male and female greater prairie-chickens favor a patchwork of burned and undisturbed tallgrass prairie. Lekking males may prefer burned sites, but they often lek near patches of residual tallgrass, presumably because these patches provide escape cover. Likewise, breeding females avoid placing nests in burned areas until the areas have recovered sufficiently to provide concealment, but those females whose clutches hatch typically move broods into a recently burned area, perhaps because small chicks can better maneuver through its more open understory, but

Table 1. Productivity of greater prairie-chicken nests on burned and unburned tallgrass prairie, Osage County, Oklahoma, 1998–2000. Statistical tests for differences between means include estimates of the effect size (d). Although the difference appears to be associated with burning of prairie, the effect is more likely related to when the nest was initiated (see text). Even so, because a much higher proportion of later nests are in burned prairie (Figure 3), they are necessarily less productive.

Variable	Burned	Unburned
Clutch size		
Sample size	13	53
Mean	9.62	11.56
Standard deviation	1.50	2.35
Standard error	0.42	0.32
ANOVA results	$F_{1,62} = 8.09$, $P < 0.01$, $d = 0.88$	
Number of fledglings		
Sample size	4	20
Mean	6.50	11.20
Standard deviation	2.08	2.48
Standard error	1.04	0.55
ANOVA results	$F_{1,22} = 12.44$, $P < 0.002$, $d = 1.93$	

more likely because recent burns have elevated invertebrate biomass on which chicks depend (Boyd and Bidwell 2001, Hagen et al. 2005).

Fuhlendorf and Engle (2004) noted that rangelands, including the prairie, have long been managed with the objective of reducing inherent landscape heterogeneity. In the tallgrass prairie, such management includes suppressing natural fires, spreading cattle grazing evenly, and setting numerous spring fires, enough that much of the habitat is burned annually. In concert, then, this effort has greatly minimized the extent of unburned tallgrass prairie. But the greater prairie-chicken prefers tallgrass prairie, of at least 1 y residual growth for nesting. As a result, management practices will need to change if we hope to conserve viable populations of this species.

A key change involves the timing and extent of fires and the associated extent of cattle grazing. Cattle gain weight more quickly when foraging on recently burned prairie (Zimmerman 1997), a result of increased forage quality and primary productivity; therefore, ranchers have an economic incentive to continue with spring burns. Yet such burns need not cover vast areas nor affect the same areas year after year. A rotation of smaller burns (and their associated grazing pressure)—the basic idea of patch burning (Johnson 1997, Fuhlendorf and Engle 2004)—would create the patchwork of burned and unburned prairie necessary for the greater prairie-chicken.

Regarding the prairie-chicken specifically, we encourage future researchers to focus on landscape-level patterns of nest and brood success on burned versus unburned prairie. For example, what ratio of burned to unburned prairie is ideal for the greater prairie-chicken? Does this ratio change with the level of habitat fragmentation? Is there a temporal pattern to burning that best suits the species? These questions could be answered with field experiments established with

the cooperation of landowners, experiments in which the extent and timing of burned plots are controlled.

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FIRE EFFECTS ON LESSER PRAIRIE-CHICKEN HABITAT

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ABSTRACT

Fire has been suppressed over vast areas of prairie and shrublands in the central and southern Great Plains. Fire suppression and habitat fragmentation promote a negative feedback mechanism that continues unabated on the contemporary landscape. The results have been negative for native plant communities and wildlife that are habitat specialists, such as the lesser prairie-chicken (*Tympanuchus pallidicinctus*). The lesser prairie-chicken has a narrowly defined habitat niche, and fire suppression and habitat fragmentation have contributed to its decline. Fire may have a positive or negative effect on specific habitat elements depending on the pre-burn condition of the element and the timing (season) of the burn. Habitat elements include lekking or booming grounds, nesting cover, brood rearing cover, and protective cover. Fire affects habitat structure for 3 y or less before the structure returns to pre-burn conditions. Historical accounts (ca. 1840) of sand shinnery oak (*Quercus havardii*) structure on the landscape are similar to a 2-y burn frequency. The juxtaposition and pattern of fire on the landscape and its effects on lesser prairie-chicken habitat are also related to herbivory and thus fire cannot be evaluated without an understanding of the fire–herbivory interaction, also known as patch burning–patch grazing or rotational grazing without fences.

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RESTORING THE FIRE–GRAZING INTERACTION FOR GRASSLAND CONSERVATION AND MANAGEMENT

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ABSTRACT

It is well recognized that fire and grazing were common features of pre–European settlement grasslands and may be important in maintaining grassland structure and function. Most studies of grazing and fire focus on binary (yes or no; a few combining fire and grazing) treatment designs on relatively small uniform experimental units. This approach of uniform treatment application is incapable of explaining complex patterns that are central to the ecological interaction of fire and grazing within the context of a dynamic landscape. Our recent research at the Tallgrass Prairie Preserve suggests that fire and grazing are interactive and can lead to a shifting mosaic landscape that is regulated through a series of positive and negative feedbacks. Unburned prairie has a high probability of fire and a low probability of selection by grazing animals. When a fire occurs on a portion of the landscape, the probability of grazing increases on that portion and animals tend to congregate and graze the burned area heavily. This fire followed by focal grazing changes the community structure and lowers the probability of fires. Grazing animals will continue to use the burned area until a new burned area becomes available; they then switch to the new burned area, allowing the previously burned area to recover. The result is a shifting mosaic of patches across the landscape, driven by the interactive influence of fire and grazing. Thus, characteristics of patches are dictated by time since focal disturbance such that the landscape contains plant communities in patches that have been recently burned and focally grazed and also patches that have not been disturbed for several years. We developed this fire–grazing model into a management model capable of enhancing biodiversity while maintaining production of domestic livestock.

Citation: Fuhlendorf, S.D., and D.M. Engle. 2007. Restoring the fire–grazing interaction for grassland conservation and management [abstract]. Page 157 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

PATCH BURNING AS A HETEROGENEITY APPROACH TO RANGELAND MANAGEMENT

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ABSTRACT

Patch burning is a novel approach to rangeland management that has potential to enhance biological diversity while maintaining livestock production on rangelands. Its foundation is based upon a long history of herbivory by large ungulates and the fire–grazing interaction on native rangelands. Traditional rangeland management generally attempts to minimize inherent rangeland heterogeneity. Patch burning, which uses the fire–grazing interaction to increase spatial heterogeneity of rangelands, allows free selection by large ungulates among burned and unburned patches within a landscape unrestricted by fencing. Because ungulates prefer to graze forage plants within recently burned patches, intense grazing moves across the landscape in the pattern created by burning patches. The result is a shifting mosaic of patches at different stages of recovery from burning and grazing disturbance. We suggest that emphasis on a heterogeneity-based approach to managing grasslands is a viable framework for conservation and a potentially useful tool in wildland fire management.

keywords: fire–grazing interaction, fuel management, heterogeneity, rangeland management.

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INTRODUCTION

Before European settlement of North America, fire and grazing by native ungulates interacted to create a dynamic and heterogeneous landscape. As individual disturbances, fire and grazing have different effects on ecosystems, but when considered as a dynamic, landscape-level interaction, their influences become more pronounced and synergistic. Grazing by bison (*Bison bison*) and other herbivores across the pre-European settlement landscape influenced the structure and function of many grasslands (Knapp et al. 1999), but the recent view is that the distribution of spatially unrestricted large ungulates within the landscape was not uniform and interacted with patterns of heterogeneity created by fire (Kay 1998; Fuhlendorf and Engle 2001, 2004). This becomes all the more evident when considering habitat requirements of habitat specialists such as grassland birds that evolved with bison grazing (Figure 1; Knopf 1996).

Managing grazing animals typically focuses on stocking rate and grazing systems with emphasis on uniform distribution of grazing disturbance (Holechek et al. 1998). As a result, most rangeland management practices (i.e., brush and weed control, fencing, etc.) have been implemented to promote livestock production by increasing dominance of a key forage species and maintaining uniform grazing patterns (Fuhlendorf and Engle 2001). Spatially variable management practices have been recommended only for attracting livestock to lightly used or unused areas with the objective

of reducing spatial heterogeneity and increasing harvest efficiency (Hooper et al. 1969, Samuel et al. 1980, Vallentine 1989, Holechek et al. 1998). Fire management has followed a similar template, with fire application usually applied to reduce forage heterogeneity and to encourage uniform distribution of livestock grazing. Without patch grazing, frequent burning as practiced on ranches in the tallgrass prairie region reduces plant diversity and increases homogeneity of tallgrass prairie (Collins 1992). A paucity of research has evaluated the interactive effects of fire and grazing on heterogeneity. Here, we present an overview of an approach to managing rangelands that promotes heterogeneity by coupling fire and grazing in a spatially dynamic fashion that, together with implications for livestock production and wildlife habitat, might be of value in grassland ecosystem management.

PATCH BURNING AS AN ALTERNATIVE MANAGEMENT APPROACH

Patch burning is an alternative management approach that uses a fire–grazing interaction to produce a shifting mosaic landscape. Grassland patches that have been not burned in several years have an increased probability of fire and a decreased probability of grazing (Figure 2; Fuhlendorf and Engle 2004). Conversely, grassland patches that have burned recently have an increased probability of grazing because forage quality is high and a decreased probability of burning again until sufficient fuel accumulates. When a patch long excluded from fire is burned, patch preference by large herbivores shifts to the most recent

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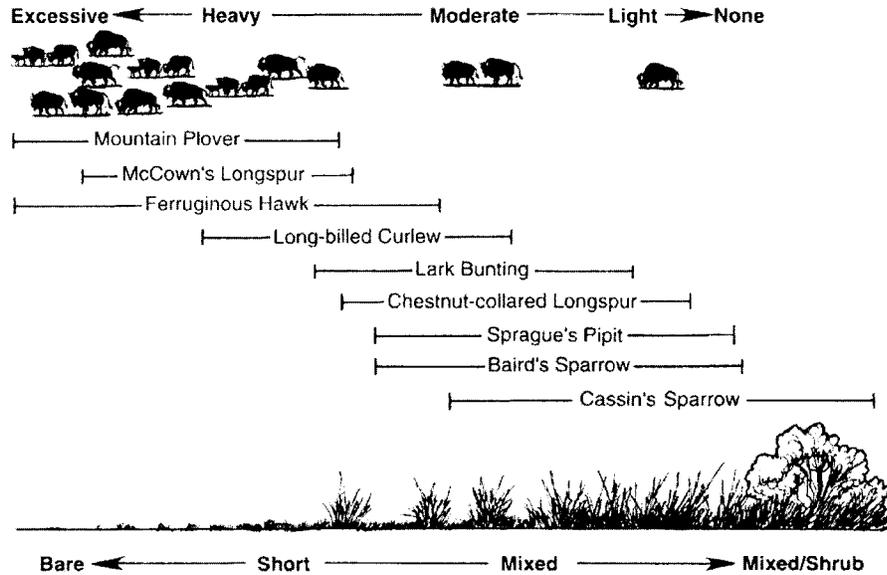


Fig. 1. The response of a shortgrass bird community to change in bison grazing intensity and vegetation structure. Different species have different habitat requirements so grazing can have a positive effect, negative effect, or no effect on different species of birds. From *Prairie Conservation* by Fred B. Samson and Fritz L. Knopf, eds. Copyright © 1996 by Island Press. Reproduced by permission of Island Press, Washington, D.C.

burn patch, and previously burned and grazed patches begin to recover. The result is a shifting mosaic landscape with diverse patches in various stages of recovery from burning and grazing, and the location of various patch types changes annually. Thus, spatiotemporal variability among patches is high, but, from a landscape perspective, stability is high because the landscape always includes a diversity of patches differing in disturbance history.

A Case Study

In 1989, The Nature Conservancy purchased the 11,800-ha Barnard Ranch in north-central Osage County, Oklahoma, and renamed it the Tallgrass Prairie Preserve. Since then, additional land acquisitions and leases have increased the preserve area to 15,600 ha. A spatially and seasonally variable prescribed burning program was initiated in March 1993, and 300

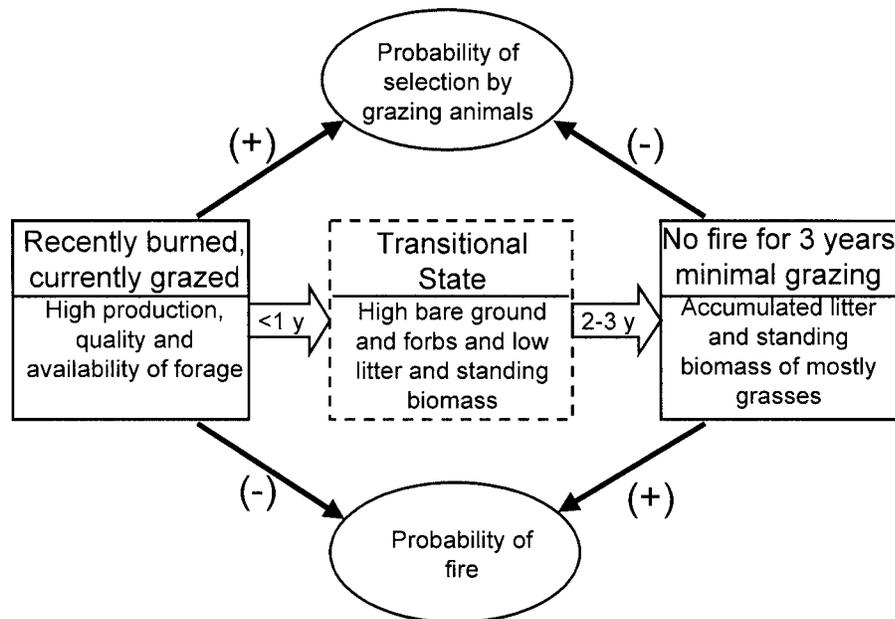


Fig. 2. A conceptual model demonstrating the dynamics of a patch within a shifting mosaic landscape where each patch is experiencing similar but out-of-phase dynamics. Ovals represent the primary drivers (fire and grazing), and squares represent the ecosystem states within a single patch as a function of time since focal disturbance. Solid arrows indicate positive (+) and negative (-) feedbacks in which plant community structure is influencing the probability of fire and grazing (Fuhlendorf and Engle 2004). Copyright © 2004 British Ecological Society. From *Journal of Applied Ecology*, by S.D. Fuhlendorf and D.M. Engle, "Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie." Reprinted by permission of Blackwell Publishing Ltd.

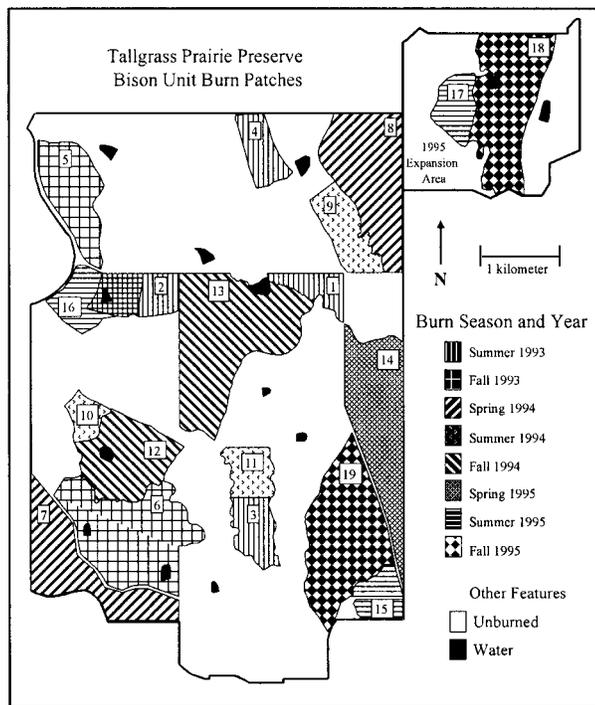


Fig. 3. Prescribed fires on the Tallgrass Prairie Preserve, Oklahoma, during 1993–1995. Numbers refer to the order in which burns were conducted. Bison within this area were allowed unrestricted selection of the landscape and primarily selected the most recently burned patches, which resulted in a shifting mosaic of patch types that were dependent upon time since fire (Coppedge and Shaw 1998). Copyright © 1998 Society for Range Management. From *Rangeland Ecology and Management*, by B.R. Coppedge and J.H. Shaw. Reprinted by permission of the authors and Alliance Communications Group, a division of Allen Press, Inc.

bison were introduced to a 1,973-ha portion of the preserve, the fire–bison unit, in October 1993. The fire–bison unit has systematically expanded with bison herd growth and now supports 2,400 bison on 8,500 ha. The fire regime within the fire–bison unit consists of 80% dormant-season (40% fall and 40% late spring) and 20% growing-season prescribed fires conducted in a random pattern intended to mimic pre–European settlement burn frequency and season. Prescribed burns are conducted on variably sized patches under a range of fuel and weather conditions with a 3-y return interval (Hamilton 1996).

The randomly located burn patches within the bison unit at the Tallgrass Prairie Preserve (Figure 3; Coppedge and Shaw 1998) have created a mosaic of patches grazed at different intensities by the bison herd. Even though stocking rate for the entire bison enclosure is moderate (6–7 ha/animal unit) (Coppedge et al. 1998a), bison use recently burned patches heavily and use unburned patches lightly (Coppedge and Shaw 1998). The result is differential succession among patches, just as other studies have predicted (Coppedge and Shaw 1998, Coppedge et al. 1998a). The effect of patch fire history on bison selectivity increases the landscape structural diversity. Forbs, the primary contributors to plant diversity in tallgrass prairie

(Howe 1994, Collins and Glenn 1995), increase within recently burned patches because bison are strongly graminivorous (Coppedge et al. 1998b).

APPLYING THE PATCH-BURNING APPROACH TO RANGELAND MANAGEMENT

A fundamental idea behind patch burning is that prescribed fire can be conducted without sacrificing stocking rate of herbivores in a management unit. Conventional wisdom for planning prescribed fires on rangelands often suggests that grazing be deferred or stocking rate reduced to allow fine fuel accumulation that will produce desired fire prescriptions (Vallentine 1989). Patch burning in conjunction with moderate stocking rate allows fuel accumulation without the economic burden of reduced stocking. Focal grazing in recently burned patches greatly reduces herbivory in adjacent patches, which allows homogeneous fuel accumulation throughout unburned patches.

Fuels and Fire Management

Patch burning might mitigate wildfire risk across the landscape by modifying grassland fuel load and by moderating fire behavior. Prescribed burning to reduce fuels is most cost-effective when subsequent fuel accumulation is limited (Fernandes and Botelho 2003). In a fire–grazing interaction, grazing pressure is focused in recently burned patches, preventing fine fuel accumulation in those patches for at least a year following burning while simultaneously allowing fuel accumulation in adjacent unburned patches. Patches recently burned and intensively grazed contain sparse fuel and function as effective firebreaks. When compared to traditional management, litter cover is less in patch-burned landscapes in all patches except those unburned the longest (Figure 4).

Strategic placement and orientation of patches within a landscape context could further enhance the effectiveness of patch burning as a fuel management tool (Finney 2001). Orienting recently burned and grazed patches that act as firebreaks perpendicular to the direction of most probable wildfire spread could reduce fire intensity and rate of spread by interrupting headfires and forcing fire to spread less intensely as backfires and flankfires (Figure 5).

Biodiversity and Biological Conservation

Birds are considered ecological indicators of ecosystem health because of their diverse habitat requirements (Bradford et al. 1998, Canterbury et al. 2000). Indeed, North American grassland birds exhibit large variability in habitat selection, ranging from preference for highly disturbed areas with minimal vegetation cover to preference for undisturbed areas with abundant vegetation litter (Wiens 1974). It is not surprising, then, that North American grassland birds are declining rapidly (Knopf 1994) on rangelands where “good” management implies moderate disturbance distributed

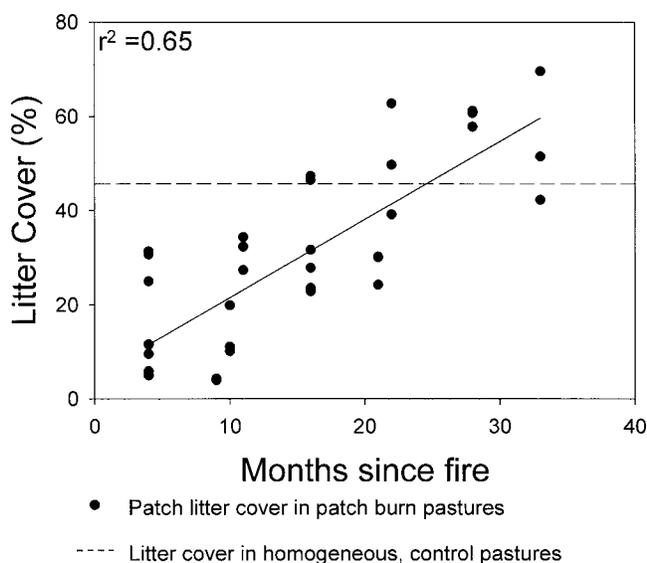


Fig. 4. Litter accumulation in pastures managed traditionally for homogeneity and in pastures managed for heterogeneity with patch burning. Litter cover is minimal in recently burned patches and recovers slowly in recently burned patches that attract intense grazing.

evenly across the landscape, precluding both heavily disturbed areas and undisturbed areas. Similar to the Tallgrass Prairie Preserve, implementation of a fire–grazing interaction on rangelands with cattle as surrogates for bison creates a shifting mosaic of vegetation types that provide diverse habitats (Fuhlendorf and Engle 2004, Townsend 2004, Fuhlendorf et al. 2006). Patch-burned landscapes provide suitable habitat for bird habitat specialists that require abundant litter and tall vegetation structure in patches not recently burned and for habitat specialists that require bare ground and open vegetation structure (Harrell 2004).

Patch burning followed by intense forage use by either bison or cattle for 1 y has not resulted in resource degradation but, rather, produces brief pulses of early seral vegetation across the landscape coinciding with recently burned patches (Fuhlendorf and Engle 2004). Following intense forage use of the burned patches, ruderal plant species increase in these patches, but within 3 y tallgrass species return to pre-disturbance dominance (Coppedge et al. 1998a, Fuhlendorf and Engle 2004). This differs from patch degradation associated with intense, recurrent grazing by livestock of preferred areas under continuous grazing systems (Fuls 1992). In patch burning, burned and intensely grazed patches are spatially dynamic, not static, so that patches that experience focal fire and grazing disturbance are rotated across the landscape over years.

Herbivore Productivity

Domestic cattle respond to patch burning management by grazing in recently burned patches 75% of the time, and cattle productivity has not declined compared to traditional range management (Fuhlendorf and Engle 2004). Similar to cattle, managing for het-

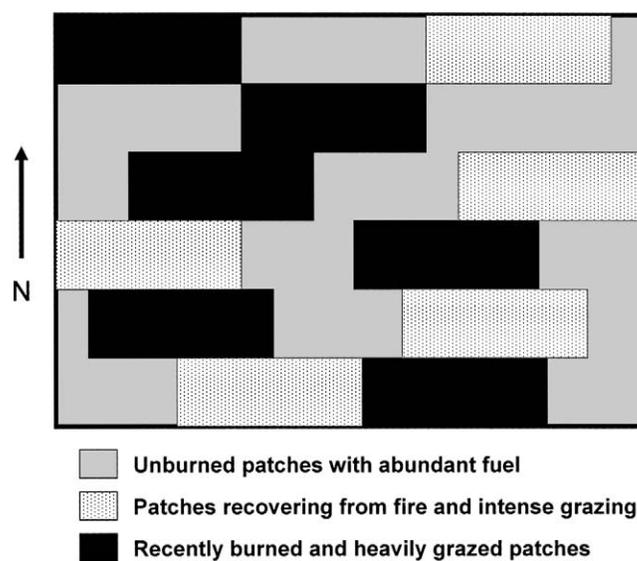


Fig. 5. Conceptual diagram of a landscape managed with patch burning to reduce wildfire potential by interrupting fire spread across a grassland landscape in which fuel beds are characteristically composed of continuous, fine fuel.

erogeneity through patch burning followed by heavy grazing has not negatively influenced bison production on the Tallgrass Prairie Preserve. In fact, bison have maintained high reproductive rates without nutritional supplementation (R. Hamilton, The Nature Conservancy, personal communication), a contrast to reduced cattle performance in the absence of nutritional supplementation under traditional rangeland management (Hughes et al. 1978, McCollum and Horn 1990). By increasing patch and vegetation diversity across the landscape, greater amounts of cool-season plants were available to bison during the winter nutritional stress period (Coppedge et al. 1998b).

Invasive Species

Central U.S. grasslands are currently under threat by two highly invasive plant species. Eastern redcedar (*Juniperus virginiana*), a native tree, is fire intolerant but has spread at an exponential rate through many grasslands that have not been burned in a decade or more (Snook 1985). Sericea lespedeza (*Lespedeza cuneata*), an exotic perennial legume, is aggressively invading many North American grasslands (Brandon et al. 2004). Our early observations suggest both these invasive species can be more effectively managed with patch burning than with traditional grazing management and prescribed fire systems. Eastern redcedar might be more vulnerable to prescribed fire in a patch-burn system because of the greater fuel accumulation in unburned patches and subsequently greater fire intensity. While neither fire nor grazing alone have detrimental effects on sericea lespedeza, frequent, intense grazing that immediately follows fire in a patch-burn system appears to reduce invasion into native grasslands (Fuhlendorf and Engle 2004).

SUMMARY

Patch burning is an alternative rangeland management option that facilitates landscape heterogeneity, whereas traditional rangeland management practices promote homogeneity. We propose that a patch-burning approach will approximate the historic range of variability on grasslands like the tallgrass prairie by reinstating a fire–grazing interaction. This approach has potential to maintain greater biodiversity by increasing heterogeneity and reducing invasive species while providing rangeland managers with greater flexibility for managing patterns of grazing and fire.

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RESTORING HETEROGENEITY ON THE TALLGRASS PRAIRIE PRESERVE: APPLYING THE FIRE–GRAZING INTERACTION MODEL

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ABSTRACT

The interaction of fire and grazing is an important ecological process in the Great Plains grasslands of North America. The fire–grazing interaction promotes a shifting mosaic of patches that support a diverse array of grassland flora and fauna. This ecosystem variability, or heterogeneity, has been identified as critical to the maintenance of biological diversity and therefore should serve as the foundation for conservation and ecosystem management.

Landscape heterogeneity has been a primary focus of the ecological management plan at The Nature Conservancy's Tallgrass Prairie Preserve in northeastern Oklahoma. Since 1993, an expanding bison (*Bison bison*) herd has been interacting with randomly selected burn patches that reflect the historical seasonality and frequency of fire. The fire–bison interaction produces vegetation structural and compositional heterogeneity in an ever-shifting landscape patch mosaic. Research and monitoring has confirmed that this heterogeneity provides for the full array of tallgrass prairie biodiversity.

The larger-scale conservation challenge is to develop cattle management regimes that incorporate some of the same “biodiversity friendly” elements as fire–bison. To that end, The Nature Conservancy is in the sixth year of a research partnership with Oklahoma State University to investigate “patch-burning” with cattle. Results thus far are very encouraging: increased heterogeneity and biodiversity can be realized with little or no decrease in livestock production.

keywords: biodiversity, bison, *Bison bison*, cattle, fire–grazing interaction, heterogeneity, Oklahoma, tallgrass prairie.

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INTRODUCTION

Ecosystem variability, or heterogeneity, has been described by numerous authors as the root of biological diversity at all levels of ecological organization, and should therefore serve as the foundation for ecosystem and conservation management (Christensen 1997, Ostfeld et al. 1997, Wiens 1997, Fuhlendorf et al. 2006). The Great Plains of North America evolved with fire and ungulate grazing, and these two agents of natural disturbance are considered by grassland ecologists to be keystone processes of the prairie ecosystem (Axelrod 1985, Milchunas et al. 1988, Knapp et al. 1999). In addition, the interaction between grazers and fire is believed to be the primary means of achieving heterogeneity and the full range of natural variation in Great Plains grasslands (Biondini et al. 1989, Vinton et al. 1993, Steuter et al. 1995, Hartnett et al. 1996, Fuhlendorf and Engle 2001). Steuter (1986) and Fuhlendorf and Engle (2004) present conceptual models of the fire–grazing interaction that demonstrates how fire and grazing, interacting through a series of positive and negative feedback loops, result in a shifting mosaic of vegetation patches across the grassland landscape (see figure 2 in Kerby et al., *this volume*).

SITE DESCRIPTION

The Tallgrass Prairie Preserve is a 15,700-ha natural area located in northeastern Oklahoma (36°50'N, 96°25'W), which is owned and managed by The Nature Conservancy. The cornerstone property, the historic 11,800-ha Barnard Ranch, was purchased by the Conservancy in late 1989, with additional property acquisitions since that time bringing the preserve to its current size. The Barnard Ranch managed for both cow–calf and yearling cattle production and used prescribed fire on a roughly 4- to 5-y rotation. Aerial application of broadleaf herbicides on a 4- to 5-y rotation was practiced since the 1950s until the ranch was purchased by the Conservancy, and is still a common regional range management practice. However, with 755 species of vascular plants collected thus far (M.W. Palmer, Oklahoma State University, unpublished data), the preserve's botanical diversity appears to be relatively intact.

The Tallgrass Prairie Preserve is located within the southern end of the Greater Flint Hills—the largest extant intact landscape of native tallgrass prairie in North America (Knopf 1994, Steinauer and Collins 1996). The 2.0-Mha Greater Flint Hills landform runs from northeastern Oklahoma through eastern Kansas and consists of two distinct vegetation associations (Figure 1). The northwestern 1.5-Mha tallgrass prairie landscape is underlain by limestone and shale, while the southeastern 0.5 Mha of crosstimbers (tallgrass

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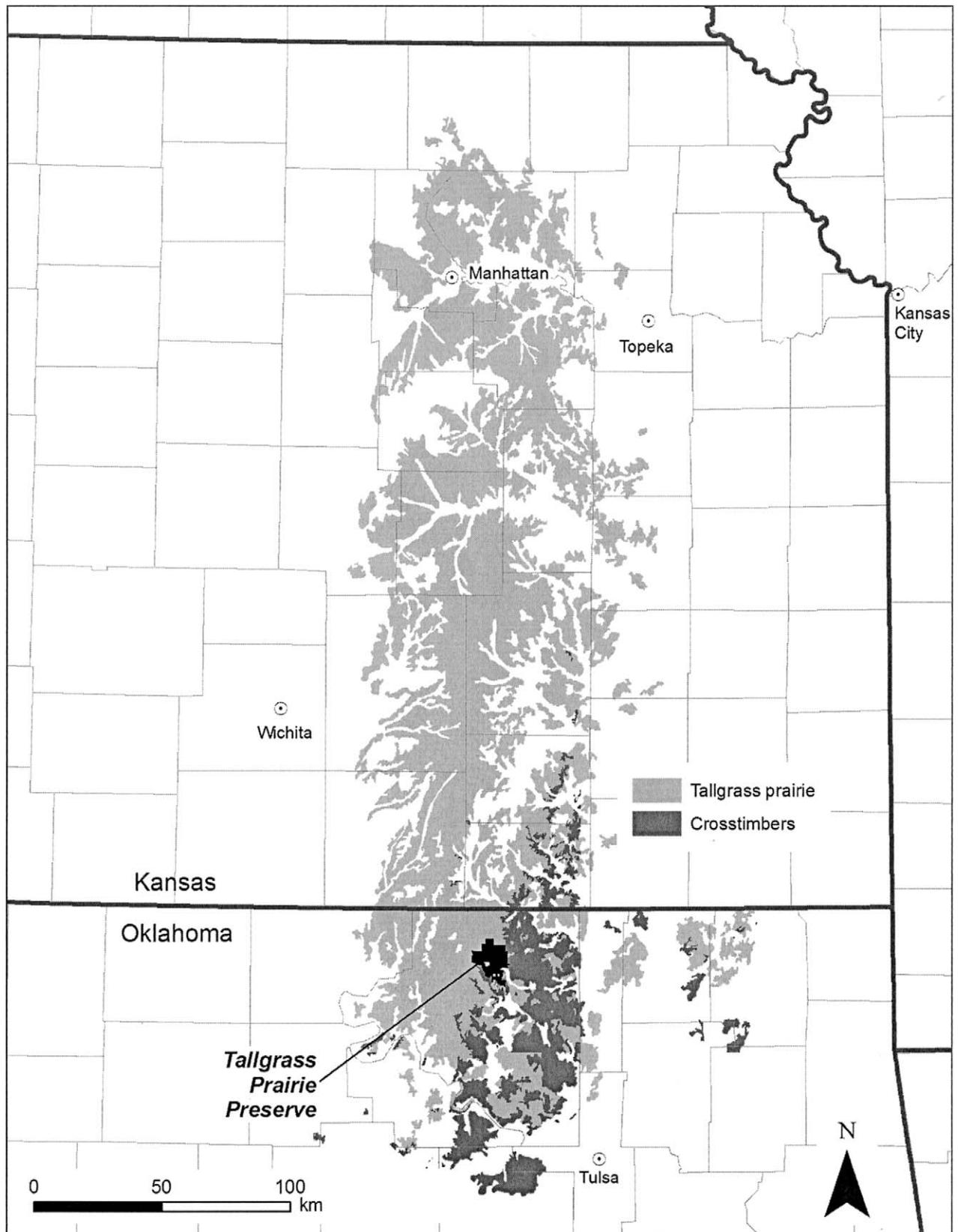


Fig. 1. The Greater Flint Hills region of eastern Kansas and northeastern Oklahoma, with the location of The Nature Conservancy's Tallgrass Prairie Preserve in Osage County, Oklahoma. Figure modified from The Nature Conservancy (2000). Reprinted by permission of The Nature Conservancy.

prairie and oak woodlands of post oak [*Quercus stellata*] and blackjack oak [*Q. marilandica*] is underlain by sandstone and shale. The Tallgrass Prairie Preserve is located on this ecotone and consists of approximately 90% grassland and 10% forest cover. Tallgrass prairie is the dominant grassland (big bluestem [*Andropogon gerardii*], Indiangrass [*Sorghastrum nutans*], composite dropseed [*Sporobolus compositus*], switchgrass [*Panicum virgatum*], and little bluestem [*Schizachyrium scoparium*]), but shortgrass prairies (dominated by grama [*Bouteloua* spp.]) occur on shallow soils and in “grazing lawns.” Most of the forested areas on the preserve are crosstimbers, but there are substantial areas of floodplain forests and mesic to xeric hardwood forests on the upper stream terraces above Sand Creek. Crosstimbers become the dominant regional vegetation type on the east side of the preserve.

Because of its rough topography and shallow, rocky soils, the Greater Flint Hills have not been cultivated as was most of the original tallgrass prairie on the continent. Ranching is the primary economic land use, with annual spring burning coupled with intensive early stocking (IES) of yearling cattle gaining in popularity over the last several decades (Launchbaugh and Owensby 1978, Smith and Owensby 1978, Vermeire and Bidwell 1998). The IES management regime involves the intensive early stocking of approximately 250 kg/head yearling cattle at twice the season-long stocking rate, but for only the first half of the growing season. The annual burn–IES range management practice results in uniform grazing use across the entire pasture, and thus a homogeneous landscape with lowered biodiversity potential. Homogenizing range practices are considered to be one of the leading sources of ecological stress in the Flint Hills (The Nature Conservancy 2000).

ECOLOGICAL MANAGEMENT PLAN

The principle of heterogeneity has been a primary consideration in the development of the ecological management plan at Tallgrass Prairie Preserve. The preserve’s operating hypothesis is that the best way to achieve The Nature Conservancy’s goal of biodiversity protection is through the maintenance of a heterogeneous mix of vegetation (habitat) patches that represent the full range of natural variation. Landscape heterogeneity will be realized through management regimes that couple fire with both native and domestic grazers. Responses of vegetation and grassland bird to management treatments are presented below in the Cattle–Fire section.

There have been several significant revisions to the ecological management plan at the Tallgrass Prairie Preserve since the original plan was published (Hamilton 1996). I describe those revisions and summarize some key research results to date.

Bison–Fire

American bison (*Bison bison*) was the dominant ungulate prior to settlement of the Great Plains (Ax-

elrod 1985, Shaw 1995, Shaw and Lee 1997). Therefore, it is intuitive that a large-scale effort to restore prairie ecosystem dynamics would utilize bison. The Tallgrass Prairie Preserve bison herd was initiated with the donation of 300 bison in the fall of 1993, which were introduced into a year-round unit of 1,960 ha. The area of the bison unit has been expanded eight times since then to keep pace with the growing herd (internal recruitment). The bison herd is still growing and currently consists of approximately 2,400 individuals in the summer season with calves, occupying a unit of 8,517 ha. The final target herd size of 2,600 bison in the summer season will be reached in 2007 and occupy a unit of 9,532 ha (Figure 2). On a year-round basis, this herd will be equivalent to 20,200 animal-unit months (AUM; 1 AUM is the forage requirement for 1 mo for a 454-kg animal unit or its equivalent).

Both the initial bison stocking rate and fire return interval were modified in summer 1999. Observations over the first 5 growing seasons (1994–1998) in the bison–fire unit confirmed that the initial bison stocking rate (forage intake target of 12.5% of annual production) was a very conservative starting point (50% below U.S. Department of Agriculture–Natural Resources Conservation Service [USDA–NRCS] recommendations). Abundant standing forage was available to the herd in all seasons, including several drought years. The revised bison forage intake target is 20% intake of annual herbage production, which averages approximately 4,300 kg/ha (Bourlier et al. 1979). When combined with an expected similar forage loss from animal trampling, total annual forage consumption will be approximately 40%, which is 20% below USDA–NRCS recommended cow–calf stocking rates for the region. The revised bison stocking rate is calculated at 2.1 AUM/ha, or 5.7 ha/AU per year (14.1 acres/AU per year).

Within the bison unit, there are no internal fences except for one imbedded pasture of 1,500 ha. However, its gates are left open at all times except for several weeks during roundup when it is used to move the herd towards the corral. The bison unit is a drive-through, “open range” situation, where visitors can drive 27 km of gravel county roads within the unit. Unfortunately, this public vehicle access within the bison unit usually results in the mortality of several animals each year from vehicle impacts.

General herd management follows the Conservancy’s guidelines for bison management (The Nature Conservancy 2002), which emphasizes minimal handling and managing for the strengths of bison as a wild species. The herd does not receive any supplemental protein or energy; however, salt with trace minerals is provided free-choice because some elements may be lacking within the bounded landscape. Water is provided by natural stream flow and small, man-made stock ponds. The entire herd is rounded up each fall and processed through a bison corral system. Since the historic large predators were extirpated over a century ago, the proper ecological stocking rate is maintained by culling surplus bison during the roundup, which are

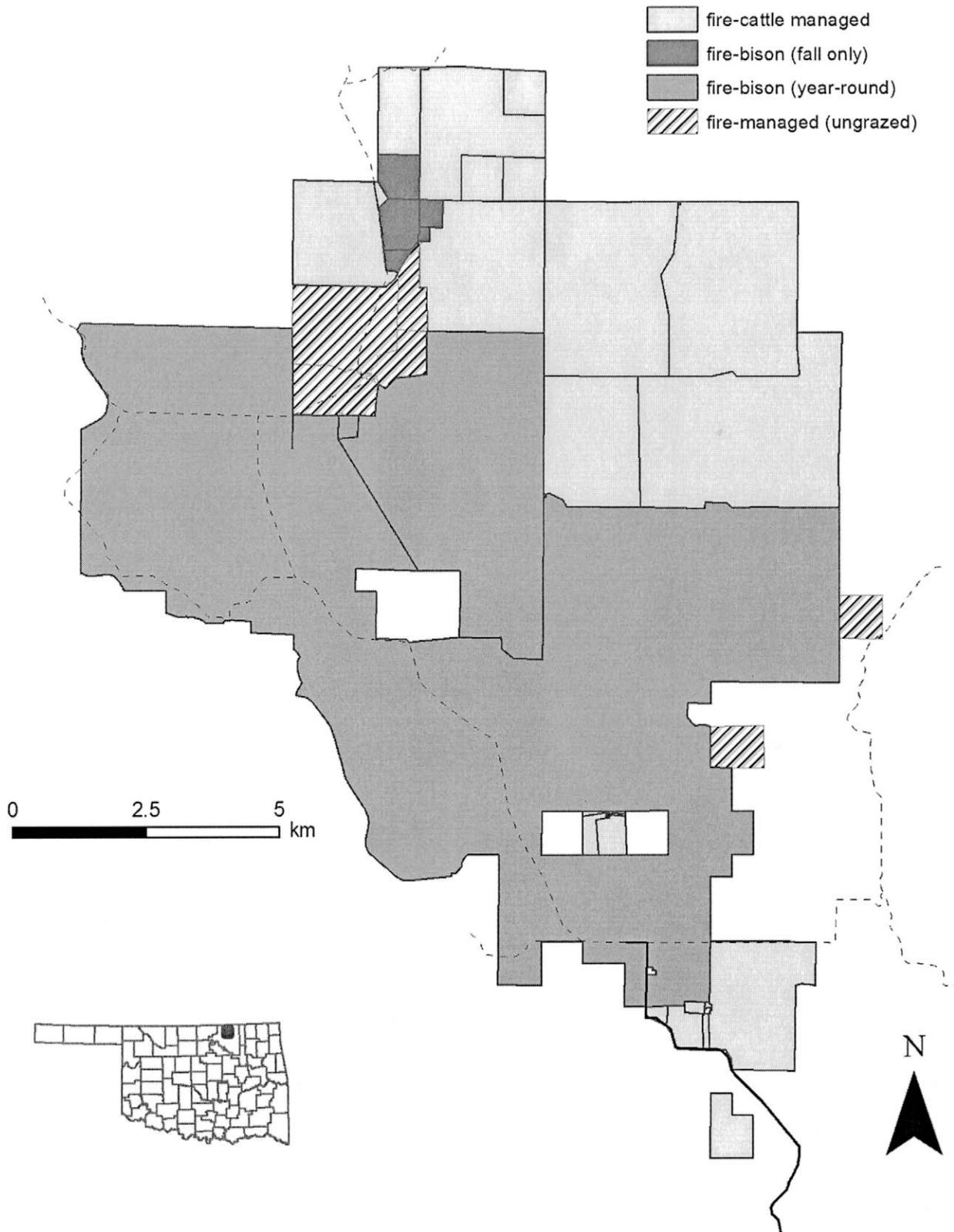


Fig. 2. Long-term management units planned for The Nature Conservancy's Tallgrass Prairie Preserve, Osage County, Oklahoma.

then sold to private bison ranchers or meat producers. Roundup is also the opportunity to apply health maintenance treatments (internal parasite control and vaccinations to prevent regional bovine diseases), with

special emphasis on preventing brucellosis and tuberculosis. All heifer calves are vaccinated to prevent brucellosis. From 1995 to 2005, the reproductive rate of the herd has averaged a 72% weaned calf crop.

All year-round the bison herd is free to move over the entire unit, with unrestricted access to all burned and unburned patches on the landscape. Burn patches are randomly selected across the unit (no fixed or permanent burn units) and reflect the historical seasonality

and frequency of fire (Figure 3). All prescribed burns are conducted by an on-site Nature Conservancy burn crew. From 1991 to 2005, 390 prescribed burns were conducted on the preserve, totaling 93,900 ha. During that same time, the Nature Conservancy burn crew has

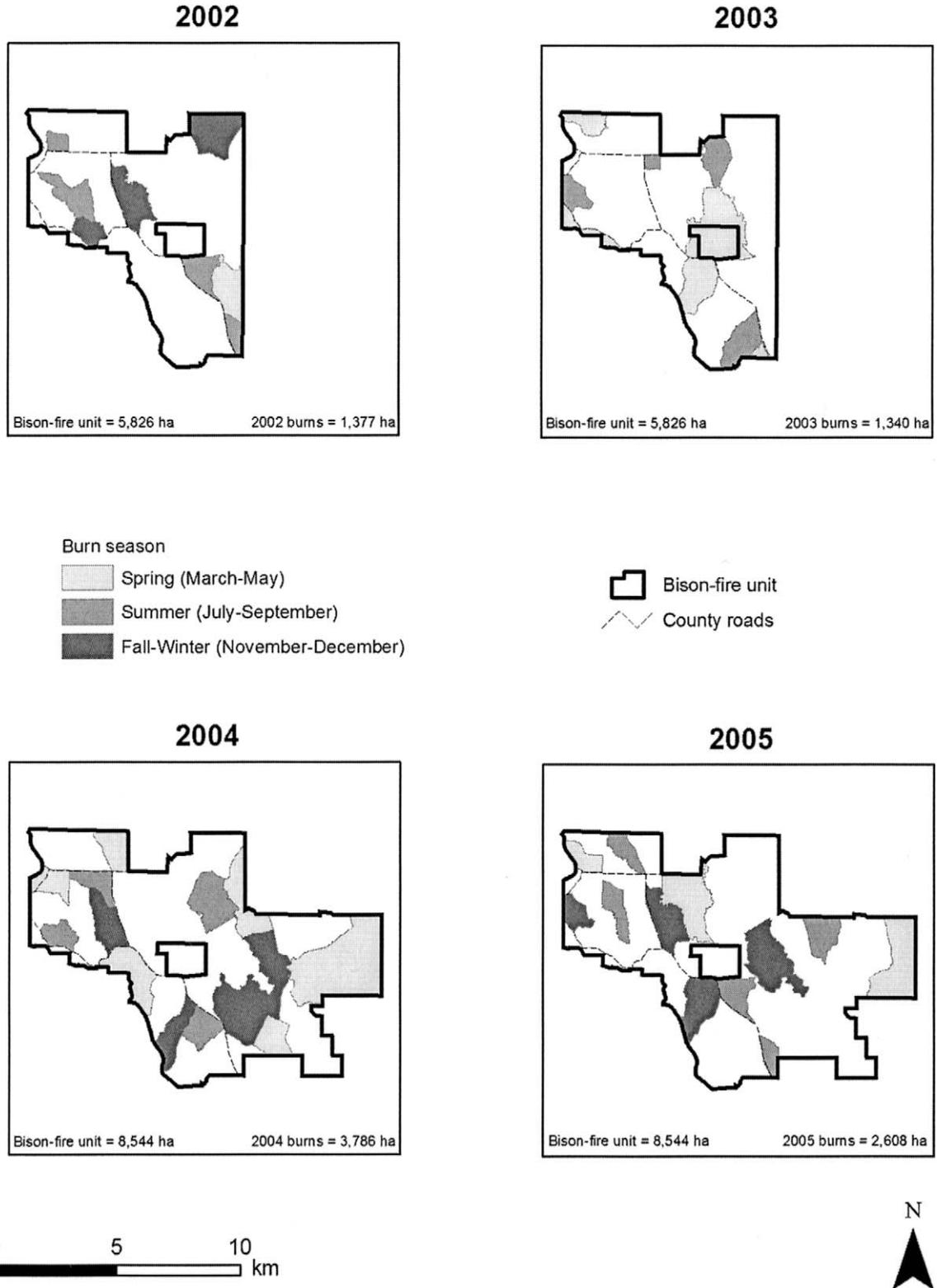


Fig. 3. Bison-fire unit burn history, 2002–2005, The Nature Conservancy's Tallgrass Prairie Preserve, Osage County, Oklahoma.

also assisted neighboring ranches in burning an additional 79,700 ha and assisted with the suppression of 52 wildfires.

The initial fire return interval in the bison unit was set at 5 y (20% of unit burned each year). However, because of the random selection system used to determine the locations of burn units, some areas were going unburned for a much longer period. Shrubs and tree seedlings were increasing in these long-unburned areas, suggesting a shorter fire return interval was needed. Therefore, the bison unit fire return interval was revised in the summer of 1999 to 3 y (33% of unit burned each year), but the seasonality of the burns will stay the same: 40% dormant spring (March–April), 20% growing season (July–September), and 40% dormant fall (November–December).

In general, the Tallgrass Prairie Preserve bison herd has reacted to fire as expected, but there have been some interesting findings from on-site research. In particular, diet selection: studies at various Great Plains sites have consistently shown bison to be strongly graminivorous, usually with graminoids making up >90% of their diets (Hartnett et al. 1997, Steuter and Hidinger 1999). However, Coppedge et al. (1998b) found that the preserve bison herd is extremely focused on grasses and sedges (*Carex* spp.) and obtain not <98% of their diet from graminoids on a year-round basis. As expected, there is a significant bison–fire interaction, with strong selection for recently burned patches during the growing season due to the increased forage quality of the regrowth following a burn (Coppedge and Shaw 1998, Coppedge et al. 1998a, Biondini et al. 1999). Interestingly, there was also a lesser but still preferential use of recent burn patches in the dormant season. Although the tallgrass prairie is dominated by warm-season grasses, cool-season sedges were accessible on burn patches in the dormant season and accounted for up to 39% and 62% of bison diets in winter and spring, respectively (Coppedge et al. 1998b). Due to this strong selection for graminoids, forbs increase dramatically within recently burned and grazed patches, but native warm-season grasses regain their dominance several years post-burn.

Cattle–Fire

The original management plan for the Tallgrass Prairie Preserve called for nearly complete elimination of cattle grazing once the bison herd covered most of the preserve (Hamilton 1996). However, the long-term bison/cattle plan was revised in late 2002 and now calls for a significant portion of the property to be kept in experimental treatments using cattle, while the “core preserve” bison–fire unit has been modestly scaled back (but will still constitute the dominant management treatment). This revision was prompted by the encouraging results from collaborative cattle “patch-burn” research initiated on 2,350 ha at the preserve in 2001 with Oklahoma State University (OSU). This project is comparing traditional intensive early stocking (IES) with patch-burn treatments. IES consists of annual spring burning coupled with stocking double

the density of yearling cattle, but only for the first half of the grazing season. The patch-burn treatments consist of burning only one-third of a cattle pasture each year (entire pasture burned after 3 y) and still stocking it at IES levels. Results thus far are very encouraging: patch-burn treatments increased heterogeneity and biodiversity with little or no decrease in livestock production when compared with traditional IES treatments (Fuhlendorf and Engle 2004). Fuhlendorf et al. (2006: figures 2, 4) found from the cattle patch-burn research conducted on the preserve that increasing the spatial and temporal heterogeneity of grassland disturbances increases vegetation variability, thus supporting a greater diversity of grassland birds.

Although the above bison–fire regime has proven to be very attractive for biodiversity conservation, it is unlikely to be very exportable to the private ranching industry that dominates the Flint Hills. Instead, a more productive strategy is to engage with conservation partners in the development of progressive cattle management systems, such as patch-burning, in an effort to impact the conservation of the Flint Hills (and other native prairie) landscapes. Starting in 2008, the next generation of patch-burn research with OSU will begin on 4,350 ha and will be testing a wider array of fire return intervals (2–4 y) and season of fire (spring-only and spring-plus-summer) using season-long grazing with yearling cattle.

ECOLOGICAL RESEARCH

In May 2004, construction of the Tallgrass Prairie Ecological Research Station was completed at the preserve headquarters in a partnership with the University of Tulsa. This modern laboratory and classroom facility will greatly enhance applied and basic ecological research on the preserve. More than three dozen research projects are currently active on the preserve, and 95 articles in scientific journals have been published. Information regarding research facilities, policies, and GIS data layers can be accessed by going to The Nature Conservancy’s website at <http://www.nature.org>, then navigating to Oklahoma, Tallgrass Prairie Preserve, then Prairie Science.

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COMPARISON OF BEHAVE: FIRE BEHAVIOR PREDICTION AND FUEL MODELING SYSTEM PREDICTIONS WITH OBSERVED FIRE BEHAVIOR VARYING BY SEASON AND FREQUENCY

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ABSTRACT

Managers are increasingly using computer models to predict prescribed fire behavior in various seasons and under different fuel conditions. Because fire behavior is related to fire effects on vegetation, validation of computer models will help understand possible outcomes and increase planning efficacy. Better predictions of fire behavior also will aid in management of risk associated with prescribed burning. We compared fire behavior predicted with BEHAVE, fire behavior and fuel modeling system, using standard and site-specific customized fuel models with observed fire behavior of strip headfires. These fires were observed in shortleaf pine (*Pinus echinata*)-dominated stands managed as pine-grassland stands for the endangered red-cockaded woodpecker (*Picoides borealis*). We evaluated the accuracy of fuel models across different seasons and fire return intervals.

Fuels in all stands tended to be heterogeneous and discontinuous, with fuel loads differing considerably between growing and dormant seasons and time since burned both in weight and composition. Fuel models varied in accuracy depending on fuel loading and season of fire. Therefore, multiple fuel models were required to more accurately characterize fire behavior across fire seasons and fire return intervals. All fuel models failed to produce accurate predictions for fireline intensity and rate of spread. All fuel models tended to overpredict fireline intensity on low-intensity fires and underpredict fireline intensity on high-intensity fires. The firing pattern we chose may have influenced the accuracy of predictions, but we were unable to detect appreciable changes in wind speed due to presumed convective influences of backing and headfires approaching each other. Additional BEHAVE fuel models for a wider range of fuel conditions show some promise in providing managers with more realistic predictions of fire behavior under variable fuel conditions. However, field validation for specific sites is imperative.

keywords: Arkansas, ecosystem restoration, fire behavior, fire ecology, fire models, *Pinus echinata*, prescribed burning, shortleaf pine.

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INTRODUCTION

Fire is a natural and dynamic force that played a major role in presettlement landscape development in both forest and rangeland communities worldwide (Christensen et al. 1981, Gill 1981, Wright and Bailey

1982, Chandler et al. 1983, Bigalke and Willan 1984, Kruger 1984, Pyne 1984, Sparks and Masters 1996). However, fire control and prevention has altered many of these fire-derived communities, often causing many plant and animal species dependent on these communities to decline or become endangered. Land managers throughout the world attempt to restore and maintain relicts of these communities by initiating prescribed fire to meet specific objectives such as endangered species management, manipulation of community composition, fire hazard reduction, wildlife habitat improvement, control of woody or invasive species, and seedbed preparation (Van Lear 1985, Waldrop et al. 1992, Wilson et al. 1995, Masters et al. 1996).

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Computer models utilizing on-site environmental data to predict common fire behavior parameters can be used before fire ignition, providing managers with insight on the probability of achieving desired objectives, possibility of escapes, and equipment required to suppress the fire front (Raybould and Roberts 1983, Andrews and Bradshaw 1990). BEHAVE is a wildland fire behavior and fuel modeling system developed by the U.S. Department of Agriculture, Forest Service to provide real-time fire behavior predictions of large-scale wildland fires or prescribed natural fires (Burgan and Rothermel 1984, Andrews 1986, Andrews and Chase 1989). However, when used with caution, BEHAVE can be utilized for fire behavior training, dispatch of crews for initial attack, and prescribed fire planning (Andrews 1986, Andrews and Chase 1989, Andrews and Bradshaw 1990).

BEHAVE is equipped with 13 standard fuel models and the capability of customizing site-specific fuel models (Andrews 1986, Andrews and Chase 1989). Standard fuel models vary according to fuel type, fuel load, and fuel structure and in their relative sensitivity to live fuel moisture. However, fuels are dynamic and describing them often requires more than one fuel model, depending on management history—particularly fire regime, the season of fire, fuel bed depth, and subsequent weather conditions. Furthermore, a standard fuel model may not adequately characterize a site; therefore, managers may need to modify a model for their particular situation. Managers must be aware of the complexity of fuels and be capable of choosing the appropriate fuel model for a given situation.

Many land managers and researchers are beginning to apply fire in different seasons and with fires of varying intensities to accomplish management objectives (Robbins and Myers 1992, Glitzenstein et al. 1995). Fire behavior characteristics may be used to predict the influence on herbaceous and woody vegetation (Engle et al. 1996, Sparks 1996, Sparks et al. 1999). Accurate fire behavior predictions can thus be used to define burning windows and potential habitat change for management of endangered species such as the red-cockaded woodpecker (*Picoides borealis*) (Sparks et al. 1999). Currently, research and experience with prescribed fire at times other than the late dormant season is limited. Therefore, managers are faced with uncertain liability risks from fire escapes, residual smoke, and uncertain management outcomes when using prescribed fire for endangered species management. However, fire behavior systems such as BEHAVE maybe useful for increasing the efficacy of prescribed fires and reducing liability (Masters and Engle 1994). Predictive fire behavior systems such as BEHAVE were developed for use in continuous fine fuels under wildfire situations. Therefore, these systems must be validated before fire management decisions are based on predicted outcomes.

Our primary objective was to evaluate the accuracy of BEHAVE's predictions by determining the most appropriate fuel model for thinned stands of shortleaf pine (*Pinus echinata*) managed as a pine-grassland community for the red-cockaded woodpeck-

er. We also used the SITE module of the Fire 1 program in BEHAVE to predict dead fine fuel moisture (Andrews 1986) for comparative purposes because some prescribed burners may not monitor fine fuel moisture. We also wanted to determine the efficacy of using a single fuel model in different seasons.

METHODS

Study Area

Our study sites were located on the Poteau Ranger District of the Ouachita National Forest (ONF) (approximately 34°54'N, 94°04'W) in Scott County of west-central Arkansas. The ONF was within the 2,280,000-ha Ouachita Mixed Forest-Meadow Provide and comprised 648,000 ha throughout the Ouachita Mountains in Arkansas and Oklahoma (Neal and Montague 1991, Bailey 1995). The Ouachita Mountains were east-to-west trending, strongly dissected, and ranged in elevation from 150 to 790 m (Fenneman 1938). Soils in the Ouachita Mountains developed from sandstone and shales and were thin and drought-prone. The climate of the area was semi-humid to humid, with hot summers and mild winters.

Our study focused on stands under active management for the endangered red-cockaded woodpecker within the 40,000-ha Pine-Bluestem Ecosystem Renewal Area (Wilson et al. 1995, Masters et al. 1996). Management consisted of thinning midstory and co-dominant pine and hardwood trees, also known as wildlife stand improvement (WSI). Dormant-season prescribed burning every 3 y followed WSI. Three-year fire intervals were the most common after WSI, but intervals varied from 1 to 4 y. We randomly chose 12 stands that had been burned previously in the dormant season between 1 and 5 times at 3-y intervals (Table 1).

Shortleaf pine was the dominant overstory tree species in all stands (Table 1). Codominant and intermediate overstory species included post oak (*Quercus stellata*), blackjack oak (*Q. marilandica*), white oak (*Q. alba*), northern red oak (*Q. rubra*), black oak (*Q. velutina*), black hickory (*Carya texana*), and mocker-nut hickory (*C. tomentosa*). Woody resprouts and shrubs (≤ 3 m) dominated the understory of these stands. The dominant understory species included poison ivy (*Toxicodendron radicans*), low-bush huckleberry (*Vaccinium pallidum*), blackberry (*Rubus* spp.), Virginia creeper (*Parthenocissus quinquefolia*), New Jersey tea (*Ceanothus americanus*), muscadine (*Vitis rotundifolia*), post oak, white oak, and shortleaf pine (Sparks 1996).

Treatments

We applied 5 treatments in a completely randomized fashion, with 2 treatments consisting of late growing-season fires, and 3 treatments of dormant-season fires. Treatments were as follows:

- 1) Late growing-season burn (G30; $n = 4$): 30 mo after previous dormant-season burn;

Table 1. Characteristics of stands used to compare BEHAVE predicted and observed fire behavior in the Ouachita Mountains of Arkansas, 1994–1996.

Stand	Treatment	Fire date	Months since last fire	Stand size (ha)	Stand elevation (m)	Slope (%)	Mean basal area (m ² /ha)	Mean height (m)	Mean DBH ^a (cm)	Mean crown length (m)	Mean crown diameter (m)	Mean canopy cover (%)
1257	D48	2 Mar 1996	48	18.2	335	9	14	17.0	32.6	9.0	7.1	68
1257	D36	2 Apr 1995	36	16.2	305	7	20	15.5	31.5	7.0	6.3	88
1257	D12	2 Mar 1996	12	16.2	305	7	20	16.0	32.5	7.0	6.4	81
1257	G30	12 Sep 1994	30	16.2	292	13	18	20.5	32.3	9.5	6.1	81
1259	G30	13 Sep 1994	30	16.2	305	9	17	21.5	33.4	10.0	6.5	72
1265	G43	14 Oct 1995	43	16.2	335	4	23	21.5	29.7	9.0	5.7	93
1274	G43	15 Oct 1995	43	17.8	305	15	26	23.0	27.6	10.0	5.2	92
1274	D36	1 Apr 1995	36	16.2	336	7	25	16.5	28.2	7.0	5.3	94
1274	D12	4 Mar 1996	12	16.2	336	7	25	16.5	28.2	7.0	5.3	84
1274	G30	10 Sep 1994	30	24.3	338	3	23	22.0	30.7	10.0	6.1	87
1289	D36	1 Apr 1995	36	16.2	335	7	17	15.0	29.1	6.0	5.4	82
1289	G30	11 Sep 1994	30	13.8	333	8	17	21.0	32.8	9.0	5.9	81
1313	D48	3 Mar 1996	48	26.7	305	8	23	15.0	26.9	7.0	4.9	84
1313	D36	31 Mar 1995	36	13.8	305	7	24	15.0	27.9	6.5	5.4	90

^a Abbreviation: DBH, diameter at breast height.

- 2) Dormant-season burn (D36; $n = 4$): 36 mo after previous dormant-season burn;
- 3) Late growing-season burn (G43; $n = 2$): 43 mo after previous dormant-season burn;
- 4) Dormant-season burn (D48; $n = 2$): 48 mo after previous dormant-season burn;
- 5) Dormant-season burn (D12; $n = 2$): 12 mo after previous dormant-season burn.

The G43 and D48 treatments differed from the G30 and D36 treatments in that experimental prescribed burns were applied after 4 growing seasons and 3 growing seasons, respectively. The G43, D48, and D12 treatments were added to test fuel model flexibility under different fuel loads.

We conducted late growing-season burns between 1200 and 1800 hours on 10–13 September 1994 and 14–15 October 1995 (Table 1). During this study, these were the earliest dates we could get a fire to readily spread in these stands; these burns were conducted well before leaf drop, and herbaceous and woody species were living. We initiated dormant-season prescribed fires between 1000 and 1800 hours on 31 March–2 April 1995 and 2–4 March 1996 (Table 1). We ignited backfires and allowed them to burn >50 m into the stand before igniting strip headfires and sampling fire behavior parameters of the strip headfires.

Fuel Sampling

We sampled fuels <1 h before burning at 3 random locations within each stand. At each location, we harvested all fuels ≤ 1.5 m in height in four to ten 0.5×0.5 -m quadrats at 5-m intervals, parallel to the fire front. We hand-separated fuels into 1-h (<0.6 cm diameter) dead, fine live fuels (all live grasses, forbs, and foliage combined <0.6 cm diameter), and 10-h (0.6–2.5 cm diameter) dead components. We weighed fuels immediately after clipping. After burning, we collected fuel residue at locations paired with pre-fire fuel samples by sampling all residual dead and live vegetation <2.5 cm in diameter to a height of 1.5 m. All fuel samples were dried at 70°C to a constant weight. Fuel moisture was calculated on a dry-weight basis. We also determined fuel moisture of 10-h fuels using standard fuels sticks and a protimeter.

We calculated fuel energy by selecting 3 random samples of dried fuels from each stand burned during the dormant season of 1995 and late growing season of 1994 ($n = 24$). We combined all fuel classes sampled for each pre-burn observation, ground samples to a fine powder, and compressed them into 1-g pellets. These pellets were then combusted in a bomb calorimeter to determine high heat of combustion.

Meteorological Data

We measured relative humidity, temperature, cloud cover, and wind speed at sunrise, 1400 hours, and sunset the day before the burn, the day of the burn, and the day after the burn. We also recorded weather observations immediately before igniting the fire, as we

Table 2. Range of fuel and weather conditions during prescribed fires on wildlife stand improvement areas in Ouachita National Forest, western Arkansas, 1994–1996.

Parameter	Treatment ^a									
	D36		G30		D48		G43		D12	
	Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.
Fuel load (kg/ha)										
Fine live fuels	73	422	320	1,890	75	330	363	1,010	81	250
1-h dead fuels	6,580	11,350	4,860	9,340	8,900	13,810	7,420	8,460	5,910	9,140
10-h dead fuels	200	1,730	156	2,510	340	1,570	77	1,700	175	1,550
Post-burn residue	3,230	6,740	2,430	9,870	5,170	9,560	3,430	6,050	5,450	8,970
Fuel moisture (%)										
Fine live fuels	45	163	92	133	114	184	86	127	133	194
1-h dead	9	45	8	28	12	28	6	15	15	24
10-h dead	6	55	6	58	28	48	8	62	19	50
Weather conditions										
Air temperature	12	24	27	31	12	16	19	26	10	19
6-m wind speed (km/h)	0	13	3	9	2	7	0	5	0	4
Relative humidity (%)	22	48	46	57	26	51	20	31	32	49
Cloud cover (%)	0	70	0	30	0	10	0	0	0	90

^a Treatment: D12, dormant-season burn 12 mo after previous dormant-season burn ($n = 6$); D36, dormant-season burn 36 mo after previous dormant-season burn ($n = 12$); D48, dormant-season burn 48 mo after previous dormant-season burn ($n = 6$); G30, late growing-season burn 30 mo after previous dormant-season burn ($n = 12$); G43, late growing-season burn 43 mo after previous dormant-season burn ($n = 6$).

observed fire behavior parameters, and immediately upon completion of the fire. We measured wind at 2 m using a totalizing anemometer. We used observed wind speeds to estimate wind speed at 6 m (Albini and Baughman 1979). We used a belt weather kit to determine other weather parameters. We verified our observations with weather data from the Poteau Ranger District Headquarters in Waldron, Arkansas, and the National Weather Service in Tulsa, Oklahoma, for Fort Smith observations (Table 2).

Stand Characteristics

We characterized the canopy species within each stand before burning. In September 1994 and in March and April, 1995, we sampled 30 points at 30-m intervals on 2–4 randomly spaced lines perpendicular to the contour. In October 1995 and March 1996, we sampled 20 locations within each stand. At each sampling location, we estimated canopy cover with a spherical densiometer (Avery 1967), tree and crown height using a clinometer, crown diameter, and diameter at breast height for the closest tree in each sampling quarter. We estimated mean tree height and the ratio of crown length to tree height and crown length to crown diameter from these observations for use in the SITE module of BEHAVE 4.1.

Fire Behavior Observations

We recorded rate of spread, flame length, flame depth, and residence time near all 3 fuel sampling locations. Before headfire ignition, we placed 3 sets of 2-m freestanding stakes, with referenced heights marked at 0.5-m intervals, at 5 m apart and perpendicular to the fire front. Three observers estimated fire behavior parameters by observing and timing the fire as the fire front passed each set of stakes, as described by Rothermel and Deeming (1980). We repeated this

procedure ≥ 2 times at 3 locations within each stand ($n \geq 18$), as logistically possible. Observed and predicted fire behavior parameters and data for each fire front observed are included in Sparks (1996).

We calculated fireline intensity by Byram's (1959) formula ($I_B = hwr$), where I_B is frontal fire intensity (kW/m); h is net heat of combustion (kJ/kg), obtained by adjusting fuel high heat of combustion for fuel moisture and heat of vaporization; w is fuel consumed (kg/m²), calculated as pre-burn fuel load minus post-burn residual fuel; and r is rate of spread (m/s). We estimated the total energy released in the active flame front, or heat per unit area (H_a ; kJ/m²), by dividing fireline intensity (kW/m) by rate of spread (m/min) (Rothermel and Deeming 1980). We determined reaction intensity (I_R ; kW/m²), or the rate of energy release per unit area of flaming zone, by dividing fireline intensity (kW/m) by flame depth (m) (Albini 1976, Alexander 1982).

Creating Custom Fuel Models

We used TSTMDL in BEHAVE to create a site-specific, static fuel model for each treatment type ($n = 5$), by adjusting the values of the Southern Rough model (fuel model 7). We used the fuels sampled in each stand to represent fuel load. We calculated depth of the fuel bed in NEWMDL and used this value in TSTMDL. After customizing fuel model 7, we used TSTMDL to test the model against observed fire behavior using environmental data we collected. We fine-tuned extinction moisture, fuel load, and fuel depth (Table 3) to produce accurate and consistent results for a variety of environmental conditions.

BEHAVE Predictions

The mathematical model used to calculate surface fire spread and intensity in BEHAVE is intended pri-

Table 3. Fuel model descriptors for standard and custom fuel models used to model fire behavior in the Ouachita Mountains, Arkansas, 1994–1996.

Model	Treatment ^a	Fuel load (Mt/ha)					Surface-to-volume ratio (1/cm)			Fuel depth (cm)	Extinction moisture (%)
		1-h	10-h	100-h	Live herba- ceous	Live woody	1-h	Live herba- ceous	Live woody		
7	All	2.54	4.20	3.37	0.00	0.83	57	6	51	76.2	40
8	All	3.37	2.25	5.61	0.00	0.00	66	6	6	6.1	30
9	All	6.54	0.93	0.34	0.00	0.00	82	6	6	6.1	25
10	All	6.74	4.49	11.23	0.00	4.49	66	6	49	30.5	25
Custom	G30	7.62	1.12	1.01	0.90	1.12	57	6	51	28.96	27
Custom	D36	8.96	1.12	0.52	0.00	0.83	57	6	51	60.9	45
Custom	G43	7.84	1.57	0.90	0.90	1.12	57	6	51	29.57	29
Custom	D12	5.60	0.67	0.45	0.00	0.83	57	6	51	24.38	25
Custom	D48	9.52	1.90	0.90	0.00	0.83	57	6	51	67.06	40

^a Treatment: D12, dormant-season burn 12 mo after previous dormant-season burn ($n = 6$); D36, dormant-season burn 36 mo after previous dormant-season burn ($n = 12$); D48, dormant-season burn 48 mo after previous dormant-season burn ($n = 6$); G30, late growing-season burn 30 mo after previous dormant-season burn ($n = 12$); G43, late growing-season burn 43 mo after previous dormant-season burn ($n = 6$).

marily for the prediction of fire behavior parameters on the flame front of a headfire carried by fine fuels (Rothermel 1983). Therefore, we only compared observed fire behavior parameters of headfires with BEHAVE predictions. We modeled fire behavior using the SITE module of the FIRE1 program in BEHAVE 4.1 (Andrews 1986). We were interested in predictions from several possible fuel models; therefore, we used our custom fuel model and standard fuel models 7, 8, 9, and 10 described by Anderson (1982) that best fit the fuel characteristics of the study area. We considered standard fuel model 5 but excluded it from consideration because the woody vegetation in our stands did not have the same characteristics of structure and a considerable amount of the dead fuel was composed of pine needles, which are somewhat volatile compared to other fuels. We used 1-h dead and fine live fuel moisture from collected fuel samples; 10-h fuel moisture from fuel sticks and protimeter readings observed on-site; we estimated 100-h fuel moisture based on 1- and 10-h fuel moisture and weather conditions. We supplied all environmental variables, stand characteristics, and weather observations for each fire subsample within a stand as prompted by the SITE module. BEHAVE predicted fireline intensity, heat per unit area, reaction intensity, flame length, and rate of spread for each fire location within all stands.

We used simple linear regression to determine if fuel models in BEHAVE were accurate predictors over the range of observed fire behavior by pairing ob-

served and predicted fire behavior parameters for each fire subsample. We tested the slope of the linear regression line (i.e., BEHAVE-predicted fire behavior versus observed fire behavior) for equality to 1, with the y-intercept forced to 0. A model was determined accurate when the slope was not significantly different from 1 at a significance level of 0.05. To determine fuel model accuracy at all levels of observed behavior, we inspected the R^2 for the model and plotted the residuals. We also validated the average accuracy of BEHAVE predictions of flame length and fireline intensity with Fisher's exact test by defining categorical variables from the fire behavior characteristics chart (Rothermel 1983).

RESULTS AND DISCUSSION

Fuel loads varied considerably, while mean fuel moisture was relatively similar across all treatments and within stands (Table 2; see Appendix A for mean fuel conditions for each fire run sampled for each stand). Fuel moisture was underpredicted considerably by fuel sticks deployed within a stand and the protimeter when compared with actual measurements, but estimates from fuel sticks and the protimeter were similar. We found that heat of combustion varied from 14,300 to 19,520 kJ/kg (Table 4) for a percent difference from the often used standard of 18,620 kJ/kg (Pyne et al. 1996) of -23.2 to 4.8 , respectively. Al-

Table 4. Mean fuel energy (kJ/kg) sampled prior to growing-season (September 1994) and dormant-season (March 1995) prescribed burns in Ouachita National Forest, Arkansas.

Stand ^a	Fire 1		Fire 2		Fire 3	
	Pre-burn	Post-burn	Pre-burn	Post-burn	Pre-burn	Post-burn
1257D	17,021.98	18,154.89	18,519.68	19,105.83	19,069.70	17,269.50
1257G	17,623.87	17,276.33	17,793.44	17,160.52	14,915.81	17,750.06
1259G	15,476.50	15,250.80	14,467.57	17,785.07	18,666.72	16,137.27
1274D	17,349.22	19,520.03	18,110.76	14,311.95	17,400.09	14,380.23
1274G	18,254.87	18,965.95	17,111.79	17,904.98	17,031.74	14,300.73
1289D	16,788.91	16,004.46	18,442.85	17,685.21	16,988.15	15,954.55
1289G	16,286.74	19,894.21	15,768.91	18,591.78	16,555.40	18,818.45
1313D	18,702.23	18,174.48	18,250.01	15,146.42	18,304.36	18,617.19

^a Abbreviations: D, dormant-season burn; G, growing-season burn.

exander (1982) suggested that most fuels were within 10% of this figure and that the influence of that amount of variation on fireline intensity is small relative to other parameters. Thus, theoretically our actual heat of combustion on the low end should result in actual fireline intensity to be lower than predicted fuel model outputs using the standard. Weather parameters varied such that we had a good range of conditions under which to compare model performance (Table 2).

Fuels in all stands tended to be heterogeneous and discontinuous, with occasional exposed rock, patches of dense grass, deep pine needles, residual thinning slash, and occasional fallen snags. Fuels in the late growing-season treatments were even more heterogeneous because of patchy distribution of live understory vegetation such as green patches of panicgrass (*Panicum* spp.) basal rosettes interspersed with dense clumps of woody vegetation.

The SITE module of the FIRE1 program in BEHAVE is based on Rothermel's (1972) model and Albini's (1976) additions. It was developed for predicting fire behavior of wildland fires in relatively homogeneous, porous fuels (Rothermel 1983). The program is intended to characterize fine fuels and describe headfires (Rothermel 1983, Andrews 1986). In discontinuous and heterogeneous fuels, the model can produce erroneous predictions (Sneeuwjagt and Frandsen 1977, Brown 1982). Brown (1982) found Rothermel's model accurately predicted rate of spread in sagebrush (*Artemisia* spp.) fuel types; however, the model produced erroneous predictions of flame length and intensity. Sneeuwjagt and Frandsen (1977) determined the model was useful in predicting fire behavior in grasslands but expressed concern about flame length and combustion zone depth inaccuracies. Therefore, part of the variation between observed and predicted fire behavior may be attributed to fuel bed continuity and relative homogeneity.

Our derivation of fireline intensity from estimates of flame length showed a considerable underestimation of fireline intensity compared with actual measurements of inputs for Byram's equation. Plots of residuals for predicted fireline intensity showed that variability of predictions increased with increasing intensity of fires (i.e., longer flame lengths) (Figure 1). Estimates of fireline intensity derived from flame lengths of headfires are problematic because of the subjectivity in estimating headfire flame lengths.

Application of Fuel Models across Seasons

Fire behavior parameters predicted with standard fuel models were constant for all treatments regardless of fuel loads and season of fire, while observed parameters varied depending on the season of the fire and fuel loads (Figure 2a–e). This constancy is linked to the static nature of the fuel models selected to best represent the stand characteristics of the study area, which were not most sensitive to live fuel moisture or loading. Fuels are dynamic, and change by seasons and with time since fire (see Sparks 1996, Sparks et al. 2002). For example, a larger proportion of the fuels

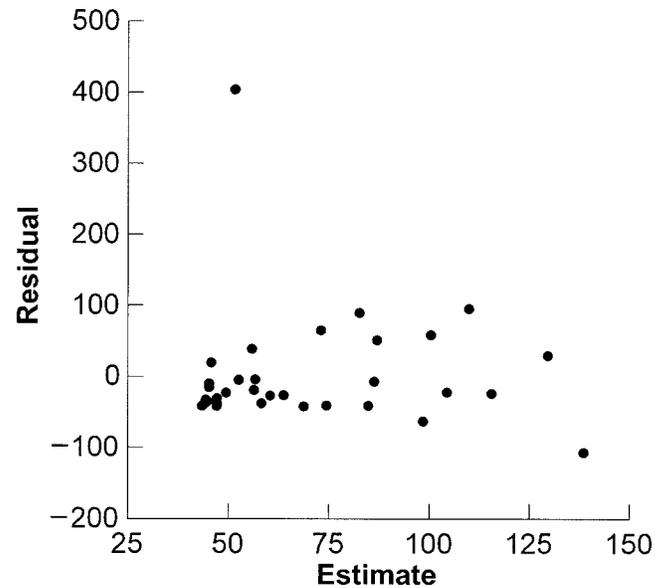


Fig. 1. Regression analysis showing plots of residuals against predicted values of fireline intensity derived from estimates of flame lengths on a series of prescribed burns on the Ouachita National Forest, Arkansas, 1994–1996.

are dormant or “cured” during dormant-season fires. Second, woody species in the understory are primarily deciduous; therefore, when they are dormant there is an increase in solar radiation and wind exposure on the fuels (Sparks 1996, Sparks et al. 2002). Further, structure of the fuel beds change with each growing season post-fire. Each fuel model is designed to function in a specific fuel type; as fuel characteristics change, so must the fuel models used to predict fire behavior. Therefore, it is essential for managers to understand the dynamics of fuels and which fuel model is appropriate for a given situation (Andrews 1986, Andrews and Chase 1989). Managers must also consider that fuel characteristics in a given stand can change through time, requiring managers to shift to different fuel models to adequately predict the fire behavior for the site for any given time. To appropriately choose a fuel model, managers should use TSTMDL in BEHAVE to validate a fuel model for their particular fuel type (Burgan and Rothermel 1984).

Accuracy of Fuel Models

We examined four standard fuel models (i.e., 7, 8, 9, and 10) that best fit the fuel characteristics of the study area and a separate fuel model customized for each treatment (Table 3). Fuel models 8 and 9 produced unrealistic predictions for all fire behavior parameters regardless of season, and because these fuel models do not consider live fuels—so in essence are dormant-season fuel models—we discarded them from further analysis. None of the fuel models, either standard or customized, produced accurate estimates of all fire behavior parameters across all treatments (Figure 2). Fuel models varied in accuracy depending on fuel loading and season of fire. Therefore, multiple fuel models were required to accurately characterize all fire

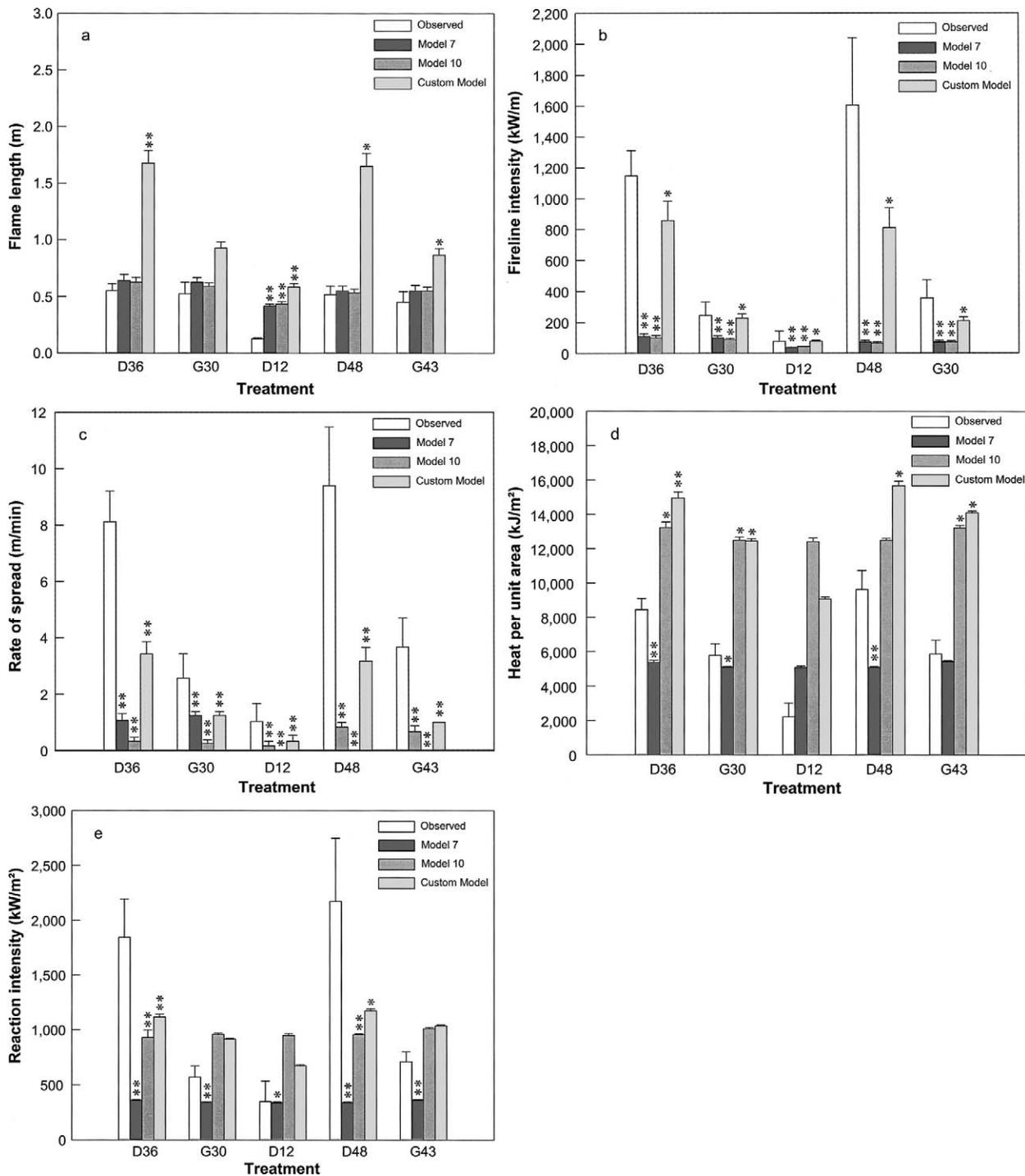


Fig. 2. Means and standard errors by treatment for (a) flame length (m), (b) fireline intensity (kW/m), (c) rate of spread (m/min), (d) heat per unit area (kJ/m²), and (e) reaction intensity (kW/m²), Ouachita Mountains, Arkansas, 1994–1996. Asterisks indicate regression results (H_0 : slope = 1, given y -intercept = 0): * = $P = 0.05$ to 0.001 , ** = $P \leq 0.001$; no asterisk indicates $P \geq 0.05$ (that the model prediction accurately represents observed fire behavior). Treatment: D12, dormant-season burn 12 mo after previous dormant-season burn ($n = 6$); D36, dormant-season burn 36 mo after previous dormant-season burn ($n = 12$); D48, dormant-season burn 48 mo after previous dormant-season burn ($n = 6$); G30, late growing-season burn 30 mo after previous dormant-season burn ($n = 12$); G43, late growing-season burn 43 mo after previous dormant-season burn ($n = 6$). Bars are +1 SE.

behavior parameters across fire seasons and fuel loads. Results on the efficacy of fuel models varied with the different statistical analyses (i.e., regression versus Fisher's exact test). For example, fuel model 7 appears to be the most accurate fuel model on average for pre-

dicting reaction intensity in the D12 treatment, but fuel model 10 and the custom fuel model are more accurate for analyzing specific fires (Figure 2e). Fuel model 7 would be effective from a management standpoint if numerous predictions were obtained throughout the

Table 5. The r^2 -values by treatment and fire behavior parameters for linear regression of observed parameters versus predicted parameters, Ouachita Mountains, Arkansas, 1994–1996.

Fire behavior parameter	Treatment ^a and model														
	D36			G30			D12			D48			G43		
	7	10	Custom	7	10	Custom	7	10	Custom	7	10	Custom	7	10	Custom
Flame length (m)	0.90	0.87	0.90	0.72	0.71	0.72	0.95	0.96	0.94	0.90	0.90	0.92	0.93	0.90	0.90
Fireline intensity (kW/m)	0.75	0.70	0.80	0.63	0.60	0.61	0.19	0.21	0.18	0.72	0.72	0.74	0.77	0.71	0.73
Rate of spread (m/min)	0.77	0.47	0.85	0.54	0.37	0.54	0.01		0.02	0.65		0.82	0.84		0.72
Heat per unit area (kJ/m ²)	0.93	0.93	0.93	0.87	0.87	0.87	0.60	0.61	0.60	0.94	0.94	0.94	0.91	0.91	0.91
Reaction intensity (kW/m ²)	0.72	0.72	0.71	0.75	0.75	0.75	0.40	0.41	0.40	0.75	0.75	0.75	0.93	0.92	0.93

^a Treatment: D12, dormant-season burn 12 mo after previous dormant-season burn ($n = 6$); D36, dormant-season burn 36 mo after previous dormant-season burn ($n = 12$); D48, dormant-season burn 48 mo after previous dormant-season burn ($n = 6$); G30, late growing-season burn 30 mo after previous dormant-season burn ($n = 12$); G43, late growing-season burn 43 mo after previous dormant-season burn ($n = 6$).

stand, but fuel model 10 and the custom fuel model would be more effective if a limited number of predictions were obtained.

Standard fuel models 7 and 10 produced accurate predictions of flame length for all treatments except the D12 treatment (Figure 2a). However, standard fuel models underestimated fireline intensity and rate of spread in all treatments except for the D12 treatment, for which the customized fuel models and fuel models 7 and 10 predicted fireline intensity similar to observed fireline intensity (Figure 2b, c). Fuel model 7 produced accurate predictions of heat per unit area more often than other models for late growing-season fires, while custom fuel models tended to overpredict heat per unit area (Figure 2d, e). Fuel model 7 was most effective at predicting reaction intensity of D12 stands, while fuel model 10 and the custom models proved more effective on average and on an individual basis for all other treatments (Figure 2e). Fuel models produced similar r^2 -values within similar treatments (Table 5).

All fuel models failed to produce accurate predictions for fireline intensity and rate of spread (Figure 2b, c). Analysis of residuals indicated that all fuel models tended to overpredict fireline intensity on low-intensity fires while underpredicting on higher-intensity fires. We found an inconsistent relationship between observed and predicted fireline intensity based on low r^2 -values for fireline intensity (Table 5). Observed rate of spread and fireline intensity may have been greater than predicted because the headfire may have been influenced by backing fires. Strip headfires were set and allowed to burn into backing fires, creating a situation similar to a ring fire, a common firing technique in the southeastern United States (Wade and Lunsford 1989). Wind speeds in the actively burning area can increase because of the convection created by ring fires (Wade and Lunsford 1989); we attempted to monitor these winds at 2 m, but were unsuccessful. Therefore, fire behavior may have been influenced by these convection winds not measured in our pre-burn weather observations. Adjustments to BEHAVE may be needed for this influence in small-scale ring fires (Masters and Engle 1994).

Because of the mountainous terrain of the region, wind speeds away from the fire were also variable in all stands, constantly shifting directions and varying

speed. We attempted to monitor this variability but were unsuccessful. As a result of live vegetation in the growing season and variable shading from the dense midstory, fuel moisture was also quite variable within stands. A portion of the large variation between observed and predicted fire parameters was a result of the inherent variation of the advancing fire front, which varies with fuel moisture, and wind speed (Brown and Davis 1973, Trollope 1984).

In the D12 treatment, BEHAVE failed to produce accurate predictions and high r^2 -values for nearly all fire behavior parameters (Table 5). Unlike other treatments, this treatment had only 1 y of fuel buildup, with fuels consisting primarily of freshly fallen conifer needles with little cured herbaceous material and hardwood leaf litter. Unweathered conifer needles, such as those found in this treatment, often act like 10-h time-lag fuels or greater (Anderson 1990). Hartford and Rothermel (1991) noted unweathered organic coatings on freshly cast conifer needles as a likely cause of slow moisture response in the 1988 fires in Yellowstone National Park.

MANAGEMENT IMPLICATIONS

If BEHAVE can accurately predict fire behavior parameters before ignition, managers can prescribe burn with more efficacy and yet reduce the risks involved. Fireline intensity, heat per unit area, and reaction intensity relate to fire effects on vegetation (Van Wagner 1973, Rothermel and Deeming 1980, Alexander 1982, Wright and Bailey 1982, Wade 1986, Engle et al. 1996). Wade (1986) recommended using fireline intensity for correlating fire behavior effects above the flame zone, reaction intensity within the flame zone, and heat per unit area for belowground effects. Flame length, which is related to fireline intensity, is a good predictor of scorch height on conifers (Van Wagner 1973). Fireline intensity and flame length are also excellent indicators of the difficulty of control, potential for escapes, and equipment required for suppression (Roussopoulos and Johnson 1975, Rothermel 1983, Pyne et al. 1996). With accurate fuel models, prescribed burn practitioners can set goals and identify parameters within which a given heat per unit area, reaction intensity, and fireline intensity will best

achieve their management objectives. Using the fire behavior characteristics chart (Rothermel 1983), practitioners can also determine the equipment required to suppress or maintain the fire fronts.

Headfires with fireline intensity <345 kW/m (flame length <1.2 m) can generally be attacked at the head by persons using hand tools (Rothermel 1983). Headfires with a fireline intensity >345 kW/m (flame length <2.5 m) are too intense for direct attack on the head by persons using hand tools and require equipment such as plows, dozers, pumpers, and retardant aircraft (Rothermel 1983). Our fires produced intensities and flame lengths in both of these categories (Figure 2a, b). In treatments D48 and D36, fuel models 10 and 7 were more accurate than custom fuel models at predicting flame length within the correct fire behavior classification ($P \leq 0.001$). However, custom fuel models were more accurate than other models at predicting fireline intensity in treatments D36 and D48 ($P \leq 0.001$).

CONCLUSIONS

BEHAVE can provide accurate predictions of fire behavior for use in defining prescribed burning windows when the proper fuel model is selected. This is particularly important when managing for endangered species because specific fire behavior parameters may be used to predict the influence on habitat variables. However, managers must proceed with caution because the appropriate fuel model varies with season of fire and fuel loading. Therefore, managers must identify or customize the appropriate fuel model. Furthermore, fuel model accuracy varies among fire behavior parameters, so managers should use more than one fuel model to predict relevant fire behavior parameters and to produce a range of fire behavior within which a specific fire may fall. Because fuels vary widely between seasons, managers should interpret predictions with caution and perform multiple analyses for each burn situation. The additional fuel models for a wider range of fuel conditions in BehavePlus 3.0.2 (Scott and Burgan 2005), plus the greater flexibility for inputs of actual values for various fuel parameters, show promise in providing managers with more realistic predictions of fire behavior under variable fuel conditions. However, field validation for specific sites is imperative.

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Appendix A. Mean fuel conditions during prescribed fires in wildlife stand improvement areas, Ouachita National Forest of western Arkansas, 1994–1996.

Stand ^a	Fire	Moisture (%)			Fuel load (kg/ha)			Post-burn residual
		1-h dead	Fine live	10-h dead	1-h dead	Fine live	10-h dead	
1257CD	1	23	184	37	10,070	294	350	5,980
	2	24	135	48	12,030	198	340	6,900
	3	24	114	39	13,805	188	1,565	6,485
1257D	1	16	120	28	9,960	200	687	4,472
	2	11	63	14	10,013	153	1,727	6,740
	3	9	45	6	10,728	160	200	6,673
1257G	1	11	101	18	9,344	1,074	2,511	6,820
	2	11	121	43	8,656	732	1,902	9,868
	3	12	121	53	9,102	1,053	1,029	8,520
1259G	1	14	101	22	8,080	1,072	1,056	5,712
	2	8	113	40	7,668	889	728	4,892
	3	10	115	31	8,888	1,416	848	6,240
1265CG	1	13	104	40	7,293	423	350	5,153
	2	14	127	36	8,460	827	857	6,053
	3	10	127	19	8,083	363	1,703	5,200
1274CG	1	10	113	51	7,613	495	77	5,747
	2	6	86	8	7,432	650	643	5,627
	3	11	110	17	7,527	1,007	703	3,433
1274D	1	12	128	26	9,247	73	1,080	3,993
	2	15	117	15	8,413	307	1,140	5,147
	3	16	120	14	6,767	283	577	6,107
1274G	1	15	105	23	7,896	870	1,212	6,256
	2	25	117	29	6,136	942	1,573	4,564
	3	8	102	11	4,864	816	280	2,432
1289D	1	22	143	29	8,180	220	1,380	4,987
	2	45	117	55	6,580	263	963	3,273
	3	20	163	45	8,260	380	1,113	3,233
1289G	1	28	92	29	8,492	1,018	1,752	5,304
	2	12	94	40	5,940	1,227	883	4,227
	3	24	100	58	5,592	1,892	156	6,028
1313CD	1	28	128	33	13,305	330	910	9,555
	2	21	114	28	8,895	75	960	5,165
	3	12	153	38	13,465	145	1,070	8,015
1313D	1	19	113	26	9,300	422	474	4,336
	2	18	138	33	10,180	366	1,436	5,448
	3	29	143	31	11,352	200	720	6,300

^a Abbreviations: D, dormant-season burn; G, growing-season burn.

EFFECTS OF LAND USE ON FUEL CHARACTERISTICS AND FIRE BEHAVIOR IN PINELANDS OF SOUTHWEST GEORGIA

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ABSTRACT

Understanding the relationship between land-use history, fuels, and fire behavior is critical for land management planning in fire-dependent ecosystems. This study sought to examine fuel loads, fire behavior, and hardwood stem kill on southeastern U.S. pinelands managed for wildlife habitat using frequent fire and selective timber thinning. Fuel loads and fuel composition were compared between native (longleaf pine [*Pinus palustris*]-wiregrass [*Aristida stricta*]) and old-field (previously cultivated land currently dominated by shortleaf pine [*Pinus echinata*]-loblolly pine [*Pinus taeda*]) communities, 1 and 2 y since previous burn, and two ranges of tree basal area (1.9–8.6 m²/ha and 8.7–19.1 m²/ha) in Southwest Georgia. Native pinelands had higher total fine fuel loads resulting from greater grass dominance and greater needle cast by longleaf pine. Duff and pine needles accumulated between the first and second years following fire. Dead grass loads increased with time in native pinelands but decreased in old-field pinelands, attributable to more competition with understory hardwoods in old fields. Tree basal area had a significant positive effect on total fine fuel loads only in native pinelands but otherwise was neutral. Flame length, rate of spread, and intensity of fires generally corresponded to fine fuel loads, with largest values occurring in native stands with high basal area in the second year following fire. Hardwood stems were generally top-killed if burned, regardless of cover type, but burns were more patchy on old-field lands as opposed to native pinelands and in 1-y compared to 2-y roughs. The results suggest that prescribed burning in native pinelands can achieve effective hardwood understory top-kill under a wider range of fire intervals, fire behavior, and weather conditions than in old-field pinelands. Burning in old-field pinelands requires additional attention to the adequacy of burn conditions to achieve management objectives using fire.

keywords: *Aristida stricta*, *Colinus virginianus*, fire frequency, fuel loads, Georgia, hardwood resprouts, loblolly pine, longleaf pine, northern bobwhite, old fields, *Pinus echinata*, *Pinus palustris*, *Pinus taeda*, prescribed fire, shortleaf pine, southeastern pine forest, timber management, wildlife management, wiregrass.

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INTRODUCTION

In southeastern U.S. pinelands, management for native wildlife must include frequent fire to prevent hardwood encroachment and to maintain an herbaceous understory (Stoddard 1935). Thus, achieving fuel conditions that allow fires to effectively top-kill hardwood resprouts is a high priority. Fuel loads and associated fire behavior might be significantly influenced by such factors as history of soil disturbance (Hedman et al. 2000), time since previous fire (McNab and Edwards 1976), and timber volume (Harrington and Edwards 1999).

Southeastern pinelands managed for fire-dependent wildlife may be broadly categorized as either “native” (never plowed) or “old-field” (forests developed after abandonment of agriculture) communities. These community types generally differ in species composition and structure (Hedman et al. 2000; Smith et al. 2000; Kirkman et al. 2004; Ostertag and Robertson, *this volume*) in a manner that may influence fuel characteristics and fire behavior. The recommended fire interval for sustaining native wildlife species

and maximizing plant biodiversity in southeastern native pinelands is 1–2 y (Moser and Palmer 1997, Conner et al. 2002, Glitzenstein et al. 2003). Land managers often alternate between burning at a 1- and 2-y rough (fire interval) on a given burn unit to balance the need for sufficient fuel accumulation with ability to kill young hardwood resprouts, as well as to provide a shifting mosaic of unburned areas for wildlife cover (Moser and Palmer 1997, Masters et al. 2003). Pine tree basal area (BA) also varies widely within the range considered to support the native herbaceous plant community and wildlife habitat in southeastern pinelands, from sparsely timbered savannas to approximately >20 m² BA/ha (90 ft²/acre) (Platt et al. 1988, Moser and Palmer 1997).

The effects of these different community types, roughs, and BA ranges on fuel loads, fire behavior, and hardwood stem top-kill are not well documented, but some general observations have been made by land managers and a small number of published studies. Native pinelands usually burn within a wider range of fine fuel moistures and with greater intensity than old-field pinelands, attributable to greater abundance of bunchgrasses, especially wiregrass (*Aristida stricta*) in the eastern Coastal Plain (Lindeman et al. 1997) and abundant, pyrogenic needles of longleaf pine (*Pinus palustris*) (McNab and Edwards 1976, Platt et al. 1991, Harrington and Edwards 1999). Burning 1 y fol-

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lowing fire, if previously treated with a 1- to 2-y fire return interval, may have the advantage of top-killing hardwood sprouts while they are small and more vulnerable and tends to promote grass dominance (White et al. 1991, Waldrop et al. 1992, Glitzenstein et al. 2003) if fuels are adequate to carry a fire. Burning 2 y following fire tends to provide a more continuous burn with greater intensity, attributable to further accumulation of fine dead fuels (McNab and Edwards 1976, Slocum et al. 2003).

With the BA range of such managed forests (up to about 20 m²/ha [90 ft²/acre]), relatively open stands provide more light, less root competition, and potentially greater productivity of herbs as fine fuels (Harrington and Edwards 1999, Dagley et al. 2002, Mulligan and Kirkman 2002). However, pine needle fuel increases with timber volume, potentially providing higher total 1-h fuel loads and higher fire intensities (McNab and Edwards 1976, Brockway and Outcalt 1998, Harrington and Edwards 1999). Denser stands may also result in higher fuel moisture due to shading.

The purpose of this study was to investigate the comparative effects of 1- versus 2-y roughs and varying levels of pine tree BA on fuel composition and loading, fire behavior, and hardwood stem top-kill in native versus old-field pinelands in southeastern U.S. uplands. Our approach was to compare plots representing these categories and their combinations. The investigation was limited to forests managed with 1- to 2-y fire intervals and thinning with single-tree selection during the previous few decades to maintain open stands for wildlife habitat.

STUDY AREA

The study was conducted within the Red Hills region of southern Georgia and northern Florida on the 1,222-ha Pebble Hill Plantation (PHP) near Thomasville, Georgia (30°35'N, 84°20'W) (Figure 1). This region is characterized by Coastal Plain marine-deposited sediments with loamy sand and sandy loam soils characterized by argillic (clay accumulation) subhorizons (Calhoun 1979). Mean annual temperature is 19.6°C (11.0–27.4°C monthly means) and mean annual precipitation is 1,373 mm (Southeast Regional Climate Center 2004). The growing season for this region is from early March to November (Calhoun 1979; T.E. Ostertag and K.M. Robertson, Tall Timbers Research Station [TTRS], unpublished data).

TTRS has managed PHP since 1990. PHP began as a cotton plantation in the 1820s during the settlement period. However, much of its area remained natural and was probably burned annually or biannually, in accordance with regional land-use traditions adopted from Native Americans and mimicking frequent lightning-initiated fires (Komarek 1979). From 1896 to the present, it has been managed for hunting of northern bobwhite (*Colinus virginianus*). During that time, most agricultural fields were allowed to succeed to old-field pinelands, of which the youngest used in this study were abandoned around 1930, such that old-

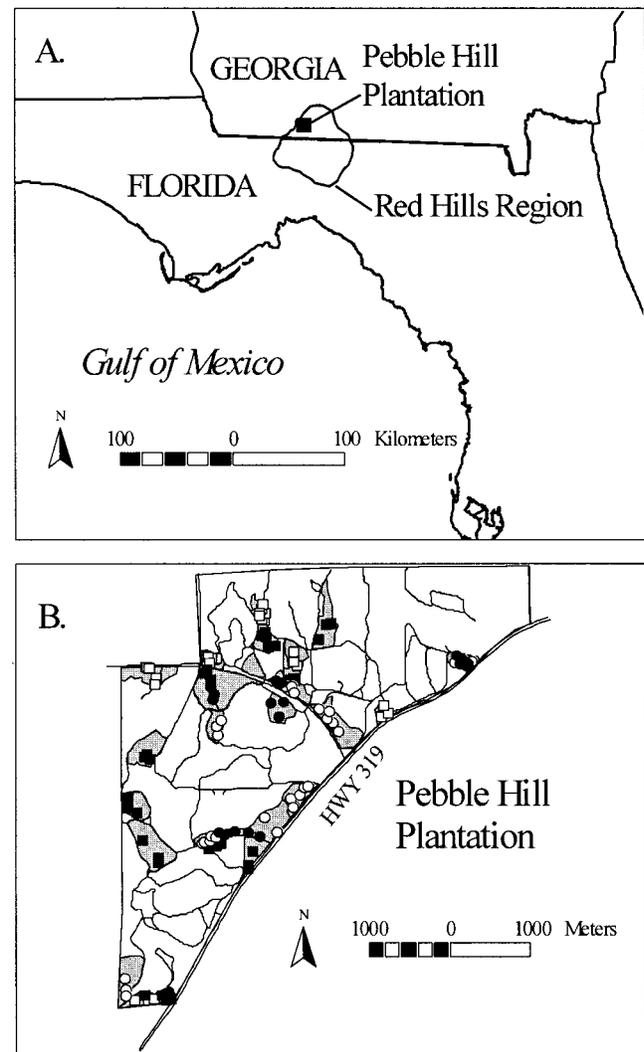


Fig. 1. (A) Red Hills region in southern Georgia and northern Florida and location of Pebble Hill Plantation (square). (B) Pebble Hill Plantation. Thin lines indicate roads, and studied burn blocks are shaded. Study plots are indicated as follows: circle = native; square = old-field; open symbol = 1 y post-burn, filled symbol = 2 y post-burn.

field pinelands studied were at least 70 y old. Northern bobwhite management has included burning of about 60–70% of both old-field and native pinelands each year (Brueckheimer 1979, Masters et al. 2003), with fires set 1 or 2 y following the previous burn. Unpaved roads traversing the property define burn units ranging in area from approximately 5 to 30 ha.

About one-third of the PHP pinelands are considered native, i.e., dominated by longleaf pine and an herbaceous layer of wiregrass, with a high diversity of other herbs and understory shrubs (Ambrose 2001; Ostertag and Robertson, *this volume*). Old-field pinelands are typically dominated by shortleaf pine (*Pinus echinata*) and/or loblolly pine (*Pinus taeda*), with an understory containing a higher proportion of forbs and woody species to grasses than native pinelands (Billings 1938; McQuilkin 1940; Oosting 1942; Hedman et al. 2000; Ostertag and Robertson, *this volume*). The overstory in both communities contains a minor component (<5% total

BA) of oaks (*Quercus* spp.), hickories (*Carya* spp.), and other hardwoods. Native pinelands on PHP are typically burned during April and May to promote flowering of wiregrass (Van Eerden 1988, Platt et al. 1991, Streng et al. 1993) and to maximize hardwood sprout top-kill (Waldrop et al. 1987, Platt et al. 1991, Glitzenstein et al. 1995), whereas old fields are burned in March when fine fuel moisture is sufficiently low to promote continuous burns. Hardwood resprouts from genetic individuals that were top-killed during the previous fire are common in both community types but usually have a higher frequency in old-field pinelands. Timber is managed to promote a multi-aged, naturally regenerating forest with BA of approximately 7–14 m²/ha (30–60 ft²/acre) at the burn-unit scale, although BA may vary widely at smaller scales.

METHODS

Field Sampling and Fire Measurements

In each of the two years of the study (2003 and 2004), the 80 upland pine burn units on PHP larger than approximately 2 ha were categorized as 1) old-field pineland with 1-y rough, 2) old-field pineland with 2-y rough, 3) native pineland with 1-y rough, and 4) native pineland with 2-y rough. Native areas had been identified in a previous study in which native forests were mapped based on the presence of wiregrass and other native indicator species (Ambrose 2001). Remaining pinelands were assumed to be old fields, based on the history of antebellum agriculture on the property and in the region (Paisley 1968, Brueckheimer 1979) and observations in the field to confirm the absence of native indicator species (Hedman et al. 2000, Kirkman et al. 2004) and dominance by shortleaf or loblolly pine.

In 2003, three burn units per community type and rough were randomly selected, and five study plot locations were randomly selected within each burn unit (60 plots total), using the Animal Movement Analysis extension (Hooge 2003) in ArcView 3.2 (Environmental Systems Research Institute 1999). In 2004, five additional burn units per community type and rough were randomly selected, and three study plot locations were randomly selected within each burn unit (60 plots total) for a total of 120 study plots, of which seven were later lost due to management accidents. Each plot was marked in the field using a piece of steel reinforcement bar. Plots were established in January and February in burn units scheduled to be burned in March or early April. Within a 2-m radius of the marker, two subplots were randomly located. At each subplot location, fuels were clipped and hand-collected to the mineral soil surface within a 0.1-m² area delineated by a square frame.

During collection, fuels were separated into duff (loose, partially decayed organic material that is difficult to differentiate), 1-h fuels (dead fuels \leq 0.6-cm thickness), 10-h fuels (dead fuels 0.7- to 2.5-cm thickness), and live herbaceous fuels. The 1-h fuels were further separated into hardwood leaf litter, pine needle

litter, dead grass, and dead forbs, vines, and hardwood seedlings combined (as these were difficult to distinguish in the field). Fuel samples were dried for 48 h at 90°C in a forced-air oven, which was found to be sufficient to achieve constant weight, and weighed to determine fuel load. Average fuel bed height was estimated at each plot. The fuel bed was considered to be the matrix of fine fuels sufficiently continuous to likely carry fire.

All trees \geq 4 cm diameter at breast height (DBH) within a 20-m radius of the plot center were identified to species and measured for DBH to determine tree BA surrounding each study plot. After sampling, tree BA was used to further categorize study plots as low BA ($<$ 8.6 m²/ha; 5.1 m²/ha average) and high BA (\geq 8.6 m²/ha; 15.0 m²/ha average), corresponding to values below and above the median measurement for all plots.

To test for correlations among fuel loads, fire behavior, and hardwood sprout top-kill, hardwood stem BA was measured within 2 mo before and 1 mo following fire at each plot. All hardwood tree or shrub stems \leq 2 cm DBH within a radius of 1–3 m from the plot center (varied to include at least 10 genets per plot) were measured for stem diameter (nearest millimeter) at 3 cm above the ground and noted for life status (live, top-killed by the fire). Species included trees and large shrubs and excluded very small-statured woody species, such as vines, running oak (*Quercus pumila*), and Darrow's blueberry (*Vaccinium darrowii*). Percent area burned within each plot was also estimated. A 2-m rod marked at 5-cm increments was laid at two random azimuths from the center of the plot, and percent area burned was estimated as the percentage of 5-cm segments intercepting burned areas.

In 2004, fire behavior variables were measured at each plot to interpret their responses to variation in fuel loads among the forest community types (native, old field), roughs (1, 2), and tree BA category (high, low). Burn prescriptions required a relative humidity (RH) of 20–40%, temperature (T) of 4–32°C, wind speed (WS) of 2–12 km/h, and 2–10 d since last rain. To assist in estimating flame length, reference markers made of metal conduit pipes marked at 0.5-m intervals were erected in a hexagonal pattern centered on each plot such that opposite pipes were 4 m apart. A strip fire was lit approximately 10 m upwind of the plot, while backing fires were at least 50 m away. Rate of spread (ROS) was measured between the first reference marker reached by the fire and the opposite pipe. Flaming combustion residence time (RT) was measured at both reference markers and averaged. Residence time was measured in seconds as the duration of flaming combustion at the base of each reference marker. Flame length (FL) was visually estimated in 25-cm increments as the flaming front passed each reference marker and estimates were averaged for each plot. During each run, wind speed was measured with a handheld weather meter (Kestrel 3000; Kestrel Design Group, Minneapolis, MN).

Analyses

Fuel loads were analyzed to determine if they differed significantly between community types, roughs,

and tree BA ranges to test for interaction effects. A multivariate analysis of variance (MANOVA) was run using community type (native or old field), rough (1 y or 2 y), and BA range (high or low) as factors, each of the measured fuel categories as response variables, and fuel load averaged between the two subplots within each plot as units of replication ($n = 113$). Also, full factorial analyses of variance (ANOVAs), Type IV sums of squares (StatSoft 2004), were used to infer the response of each fuel type to community, rough, and BA range and their interactions. ANOVAs were similarly run to test for responses in hardwood sprout BA per hectare, tree BA per hectare, and fuel bed depth to treatment levels. Correlations between tree BA, hardwood stem BA, and fuel loads in each fuel category were tested (StatSoft 2004) to interpret possible effects of forest structure and competitive interactions on fuel loads.

Using data from the 2004 plots ($n = 54$), the effects of community, year, and tree BA on ROS, FL, RT, and estimated fireline intensity (FI) and reaction intensity (RI) were tested in separate analyses using full factorial ANOVAs, Type VI sums of squares (StatSoft 2004). A similar analysis was used to test for variation among treatments in T, RH, and WS measured during burns to determine whether or not fires occurred under relatively uniform conditions among treatments.

The percentages of hardwood stems burned at the base and the subset of those top-killed were compared between community types, roughs, and tree BA ranges. Comparisons were qualitative because of the large number of plots that had 100% top-kill.

RESULTS

Trees >4 cm DBH in native pineland plots were composed of 75% longleaf pine, 14% shortleaf pine, 3% loblolly pine, and 8% hardwoods by BA. In contrast, old-field pinelands comprised 14% longleaf, 44% shortleaf, 32% loblolly, and 10% hardwoods.

Fuels differed significantly in composition and loading among community types, roughs, and tree BA ranges in the studied forests. The MANOVA testing for the effects of these factors and their interactions on fuel loads showed each factor and two of the three interactions to be highly significant (Table 1). Univariate ANOVAs revealed that each fuel type, except 10-h fuels, had a significant response to community type, rough, and/or tree BA range (Figure 2).

Native pineland sites were characterized by significantly higher 1-h fuel loads, including higher needle loads and dead grass loads, as well as higher live herb loads compared to old-field sites (Figure 2). However, native sites also had lower duff loads. The 2-y-rough plots had significantly higher duff loads, needle loads, forb-woody 1-h fuel loads, and total 1-h fuels than 1-y-rough plots (Figure 2). Plots in the high tree BA range had higher needle loads and higher total 1-h fuels overall (Figure 2). Needle loads were significantly higher in native sites than in old-field sites (Figure 2).

Table 1. Main effects and two-way interaction effects (multivariate analysis of variance) of community type (native versus old-field pineland), rough (1- versus 2-y), and tree basal area (BA) (below versus above 8.6 m²/ha) on fuel loads in multiple fuel categories in 2003 and 2004 on Pebble Hill Plantation, Georgia.

Factor	df	Pillai's trace	F	P
Community	1	0.697	45.1	<0.001
Rough	1	0.525	22.8	<0.001
Tree BA	1	0.298	8.7	<0.001
Community × Rough	1	0.218	5.7	<0.001
Community × Tree BA	1	0.232	6.2	<0.001
Rough × Tree BA	1	0.042	0.9	0.48
Residuals	107			

Tree BA was not significantly different between the two community types or between 1- and 2-y roughs (Figure 3), reflecting the timber management of these forests. In contrast, hardwood resprout BA was significantly higher in old-field pinelands and in 2-y roughs. In old-field plots, hardwood resprout BA was lower in the high tree BA category. Fuel bed height responded significantly only to community type, with highest levels in the native sites (Figure 3).

Correlations among fuel load types were found to be qualitatively the same between 1- and 2-y roughs, so results were presented separately for native and old-field pinelands and combined otherwise (Table 2). In native pinelands, tree BA was correlated with total 1-h fuel and needle litter loads, and hardwood stem BA was correlated with total 1-h fuel and hardwood leaf litter loads (Table 2). In old-field pinelands, similar relationships were evident between tree BA and needle loads and hardwood resprout BA and leaf litter loads, respectively, but we did not detect responses in total 1-h fuel loads (Table 2).

Weather variables (T, RH, WS) measured during the 2004 burns did not vary on average between community types, roughs, and tree BA ranges, with regard to main effects (Table 3). In contrast, fire behavior measurements had significant responses to certain factors (Figure 4). Fire behavior was generally more active (higher FL, ROS, FI, RI, heat per unit area) in native than in old-field plots, and more active in 2-y than in 1-y roughs in native pinelands (Figure 4). Residence times in old-field plots were approximately twice those of native plots in a given rough and tree BA range (Figure 4).

A total of 5,971 hardwood resprouts were censused in the plots (Table 4). The number of stems was lower in the 2-y-rough plots, while the average basal diameter of stems increased with time since fire (Table 4). Percentage of stems burned at the base was higher in the native than in the old-field plots and higher in the 2-y than in the 1-y rough, while differences between tree BA categories were not consistent (Table 4).

DISCUSSION

Fuel Loads

Fuel loads and corresponding fire behavior vary according to soil disturbance history, time since pre-

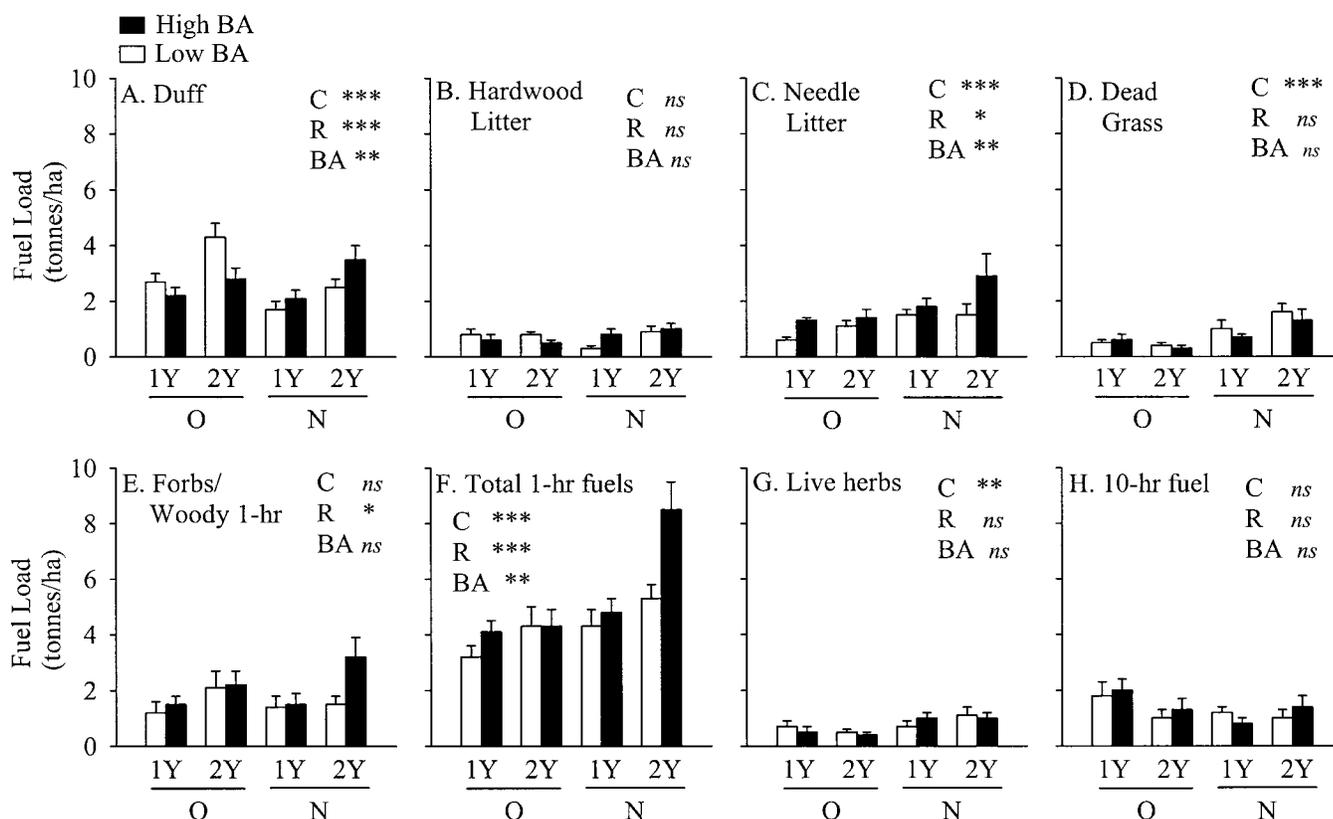


Fig. 2. Fuel loads for each fuel category in native versus old-field pinelands communities on Pebble Hill Plantation, Georgia, 2003–2004, in each basal area (BA) range (low BA = 2–8.6 m²/ha; high BA = 8.6–25 m²/ha), rough (1 y post-burn [1Y] or 2 y post-burn [2Y]), and community type (N, native pineland, O, old-field pineland). Fuel categories include (A) duff, (B) hardwood litter, (C) needle litter, (D) dead grass, (E) forbs–woody 1-h fuels, (F) total 1-h fuels, (G) live herbs, and (H) 10-h fuels. Mean and standard error for plots are given (12–17 plots/bar, 113 plots total). Land cover main effects (C = community type, R = rough, BA) from ANOVAs are also given. *ns*, not significant. * $P = 0.05–0.01$, ** $P = 0.01–0.001$, *** $P < 0.001$.

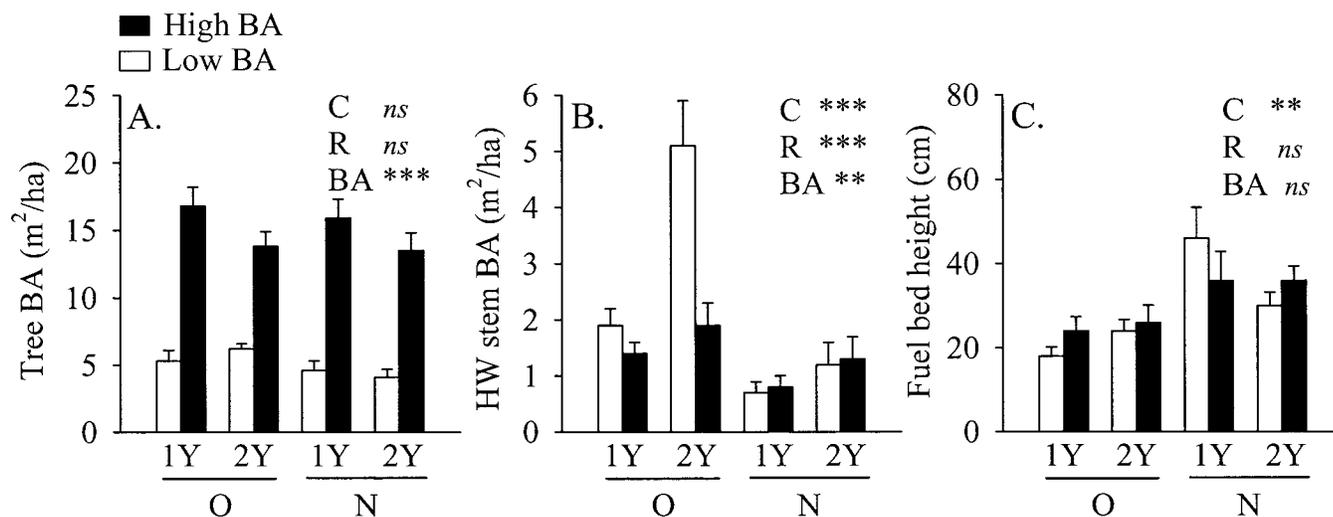


Fig. 3. Levels of (A) tree basal area (BA), (B) hardwood stem BA, and (C) fuel bed height in each BA range (low BA = 2–8.6 m²/ha; high BA = 8.6–25 m²/ha), rough (1 y post-burn [1Y] or 2 y post-burn [2Y]), and community type (N, native pineland; O, old-field pineland) in native versus old-field pinelands communities on Pebble Hill Plantation, Georgia, 2003–2004. Mean and standard error of plots are given (see Table 3 for number of plots). Land cover main effects (C = community type, R = rough, BA) from ANOVAs are also given. *ns*, not significant. * $P = 0.05–0.01$, ** $P = 0.01–0.001$, *** $P < 0.001$.

Table 2. Correlations ($\alpha = 0.05$) between pairs of variables measured at each plot in native versus old-field pineland communities on Pebble Hill Plantation, Georgia, 2003–2004.

Variable	Native plots ($n = 60$)		Old-field plots ($n = 60$)	
	Tree BA ^a	HW stem BA ^b	Tree BA	HW stem BA
HW stem BA	ns ^c		-0.38	
Duff	0.43	ns	-0.24	0.47
HW litter (1-hr)	ns	0.73	ns	0.32
Needle litter (1-hr)	0.32	ns	0.22	ns
Dead forb/woody (1-hr)	0.37	ns	ns	ns
Dead grass (1-hr)	ns	ns	ns	ns
Total 1-hr	0.44	0.33	ns	ns

^a Basal area (BA) of trees >2 cm diameter at breast height (DBH) within 10-m radius of plot center.

^b Basal area of hardwoods (HW) <2 cm DBH within the plot.

^c ns, non-significant.

vious burn, and pine tree canopy cover in the studied pinelands. These variations may be attributed in part to species composition and competitive interactions between canopy, understory, and herbaceous plants, which vary among the land uses.

Native pinelands appear to have higher rates of herbaceous vegetation productivity, particularly with regard to grasses, contributing to higher total 1-h fuel loads in that community type. Correspondingly low understory hardwood density in native pinelands is consistent with the suggestion that dominance of bunchgrasses in longleaf pine–wiregrass forests limits hardwood dominance by burning intensely under a wide range of conditions (Noss 1989, Platt et al. 1991). In turn, competition between hardwoods and herbaceous plants is kept low, resulting in higher herb productivity (Brockway et al. 1998, Harrington and Edwards 1999). Greater abundance of grasses relative to forbs and leaf litter increases horizontal continuity of standing fine fuel and presumably results in lower fine fuel moisture. An increase in grass biomass in native pinelands between the first and second years following fire also suggests that grasses are not being suppressed by the hardwood understory, in contrast to old-field pinelands, where grass fuel loads decreased in the second year while hardwood dominance rapidly increased. Also, the most dominant forbs in the old fields, namely sand blackberry (*Rubus cuneifolius*), wrinkleleaf goldenrod (*Solidago rugosa*), and hyssop-leaf thoroughwort (*Eupatorium hyssopifolium*) (Oster-

tag and Robertson, *this volume*), are large in stature and may compete with grasses in a manner similar to understory hardwoods. These stand in contrast to smaller-statured dominant forbs such as early blue violet (*Viola palmata*), pineland silkgrass (*Pityopsis aspera*), sensitive brier (*Mimosa quadrivalvis*), and oblongleaf snakeherb (*Dyschoriste oblongifolia*) in native pinelands (Kirkman et al. 2004; Ostertag and Robertson, *this volume*). The greater hardwood stem dominance in old fields likely relates to more successful establishment of hardwood tree and forb species relative to grasses following abandonment of plowed fields (Van Eerden 1988, Seamon et al. 1989, McGee 1996, Kirkman et al. 2004), creating an environment that is less flammable and more conducive to the survival of hardwoods.

Variation in pine needle litter in association with pine tree BA also contributed to differences in total 1-h fuel among land cover types. This positive association has been quantified in other studies (McNab and Edwards 1976, Harrington and Edwards 1999). Higher 1-h fuel loads in native versus old-field pinelands appear to result from differences in the dominant pine species, with longleaf pine producing significantly higher needle loads than loblolly and shortleaf pines at comparable levels of tree BA (Figure 2). Longleaf pine has longer needles than the other species and retains needles for only 2 y, compared to the 3-y needle retention time of loblolly and shortleaf pines (Mirov 1967, Landers 1991).

Within the relatively low range of tree BA studied (1.9–19.1 m²/ha [8.3–83.2 ft²/acre]), pine trees did not appear to have a negative competitive effect on grasses and forbs, judging from dead grass, forb, and live herbaceous fuel load responses to tree BA. Another study in longleaf pinelands with comparable ranges of tree BA and uneven-aged stands showed a limited response of herbaceous biomass to tree BA (Moser and Yu 1999). However, other studies of longleaf pine within this range of BA showed evidence for competitive effects of overstory trees on understory herbs (Wolters 1981, Mulligan and Kirkman 2002, Harrington et al. 2003), but those were conducted in young (<20 y), relatively densely stocked pine plantations where canopy cover may have been higher relative to BA.

The neutral effect of tree BA in old fields may relate to the negative relationship between tree BA and hardwood resprout BA found in this study, attributable to root competition or shading (Harrington and Ed-

Table 3. Means (\pm standard deviation) of weather variables during fire behavior measurements in native versus old-field pineland communities in 2004 on Pebble Hill Plantation, Georgia, 2003–2004.

Variable ^a	Old field				Native				All plots
	1-y rough		2-y rough		1-y rough		2-y rough		
	Low BA ^b	High BA	Low BA	High BA	Low BA	High BA	Low BA	High BA	
No. of plots	8	6	9	5	5	6	10	5	54
T (°C)	74 (6)	75 (3)	69 (4)	75 (6)	80 (7)	76 (9)	71 (7)	74 (5)	74 (6)
RH (%)	34 (7)	30 (8)	32 (8)	30 (10)	26 (5)	34 (11)	35 (8)	29 (7)	31 (8)
WS (km/h)	2.5 (1.0)	4.8 (1.3)	3.6 (0.8)	3.2 (1.3)	3.4 (2.3)	3.6 (1.3)	3.2 (0.9)	2.9 (0.6)	3.4 (1.3)

^a Abbreviations: RH, relative humidity; T, temperature; WS, wind speed.

^b BA, basal area.

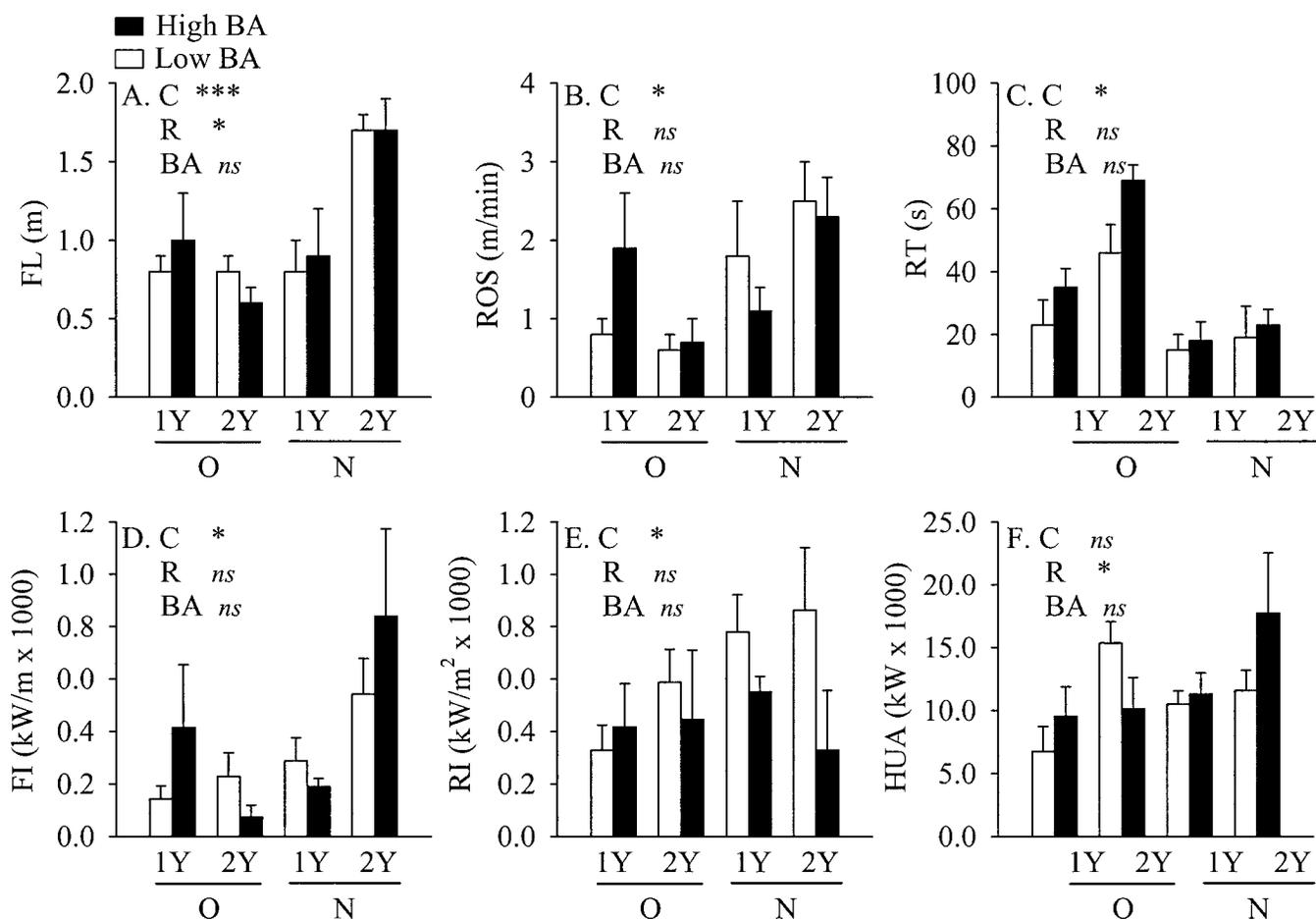


Fig. 4. Fire behavior measurements in native versus old-field pinelands communities on Pebble Hill Plantation, Georgia, 2004, in each basal area (BA) range (low BA = 2–8.6 m²/ha; high BA = 8.6–25 m²/ha), year since previous fire (1 y post-burn [1Y] or 2 y post-burn [2Y]), and community type (N, native pineland; O, old-field pineland). Mean and standard error of plots are given (see Table 3 for number of plots). (A) Flame length (FL), (B) rate of spread (ROS), (C) reaction time (RT), (D) fireline intensity (FI), (E) reaction intensity (RI), (F) heat per unit area (HUA). Land cover main effects (C = community type, R = rough, BA) from ANOVAs are also given. ns, not significant. * P = 0.05–0.01, ** P = 0.01–0.001, *** P < 0.001.

wards 1999). The release of herbs from hardwood competition may compensate for competition between herbs and pines. This could explain the positive correlation between forb 1-h fuels and pine tree BA in native pinelands. Increases in tree BA beyond the range in this study are expected at some point to have a negative effect on herbaceous cover through shading, root competition, and litter fall (Brender et al. 1976,

Masters et al. 1993, Harrington and Edwards 1999, Dagley et al. 2002, Harrington et al. 2003). Differences in crown structure suggest that longleaf pines allow a higher level of light penetration per basal area than do loblolly and shortleaf pines (Landers 1991), such that the suppression of herbs by pine canopy shading should occur at a higher level of BA in native pinelands. Correspondingly, 2005 timber cruise data on

Table 4. Mean hardwood stem (<2 cm diameter at breast height) responses to prescribed fire in native versus old-field pineland communities in June–July of 2003 and 2004 on Pebble Hill Plantation, Georgia.

Hardwood stems	Old field				Native				All plots
	1-y rough		2-y rough		1-y rough		2-y rough		
	Low BA ^a	High BA	Low BA	High BA	Low BA	High BA	Low BA	High BA	
No. plots	11	16	15	15	8	9	13	11	98
No. stems total	807	1,260	700	721	803	670	522	488	5,971
No. stems/plot	67	74	47	48	62	61	40	41	55
Stem base diameter (mm)	4.3	3.9	6.7	5.1	3.0	3.2	4.8	4.1	4.3
% burned	76.3	62.4	96.2	88.4	84.2	99.7	99.5	100.0	87.5
% top-killed	67.8	59.2	94.8	88.2	84.6	100.0	100.0	99.9	85.4
% top-killed of burned	92.6	99.3	96.5	99.2	100.0	100.0	100.0	99.9	98.0

^a BA, basal area.

PHP showed that the regression slope of canopy cover (measured with a siting tube) over BA (measured with a prism) for longleaf pine was only 60% that of shortleaf and loblolly pines (TTRS, unpublished data).

The fuel load estimates in this study may be used as input for fire behavior models such as BEHAVE series (Andrews et al. 2003). The average native pineland fuel loads and bed depths found in this study correspond closely to fuel model 2 (Rothermel 1972), while some variation in fire behavior in response to time since previous fire and tree BA is expected to occur based on our results. Old-field pinelands are better represented by a hybrid of fuel models 2 and 8 (Rothermel 1972), given that 1-h, 10-h, and live herb loads and fuel bed height are intermediate between these two models. BehavePlus 2.0 has the option of combining the two fuel models, in which case a 50% coverage of fuel models 2 and 8 is recommended for old-field pinelands. The specific fuel loads provided by this study may also be used directly.

Fire Behavior and Hardwood Stem Top-kill

Overall, variation in fire behavior among the land cover types corresponded to patterns of 1-h fuel loading, with higher levels of ROS, FL, FI, and RI in the 2-y-rough plots of native pinelands. This pattern is attributable in part to the higher rate of accumulation of 1-h fuel loads in native pinelands, for which the difference between 1- and 2-y roughs was twice that of old-field pinelands. Differences in fuel structure between native and old-field pinelands also likely had an important effect. Native pineland fine fuels were less compact, having relatively higher proportions of standing dead fuels, higher fuel bed depths, abundant pine needles that drape in the grass, and lower duff loads, and thus were better aerated, prone to drying, and likely to promote higher fire intensities, rates of spread, and lower residence times (Noss 1989, Platt et al. 1991). Exploratory correlation analyses revealed that duff loads were significantly correlated with hardwood litter and forb fuels (data not shown), which may have higher decomposition rates and a more rapid transition from litter to duff fuels (Berg and Ekbohm 1991, Prescott et al. 2004), explaining in part the higher levels of duff in old-field pinelands. Also, lower fire intensity, patchier burns, and lower flammability of broadleaf litter fuels characteristic of the old fields may contribute to lower total combustion of 1-h fuels and greater duff accumulation.

The very close association between stems burned at the base and those top-killed suggests that completeness of area burned is an accurate predictor of stem kill within 2 y following the previous fire. The difference in percentage of total stems that were burned between the first and second years following fire in old fields (68% versus 92%) is attributable to lower fuel loads in the first year, resulting in patchier burns (Slocum et al. 2003). In native pinelands, the more modest change from 92% to nearly 100% stems burned in the first versus the second years is similarly attributable to fuel load accumulation and uniformity

of the burn. However, in old fields, the higher survival rate of burned stems in the first year post-burn is attributable to lower fire intensity associated with lower fuel loads. Stems burned but not top-killed were limited to those with the largest diameters, presumably reflecting greater bark development and insulation of the vascular cambium (Hare 1965). Such survivors, which were overwhelmingly in old fields, effectively double their fire-free interval and have a higher chance of surviving subsequent burns. In turn, surviving stems increase shading, outcompete herbaceous vegetation, and shift the fuel structure to one that is more compact and less flammable.

The trend of increasing grass loads over time in native pinelands suggests that a 3- or 4-y fire-free interval may provide conditions for even more intense fires under a relatively wide range of weather conditions. However, at some length of fire-free interval the hardwoods would become less susceptible to even intense fires because of thickening of the bark (Hare 1965). In contrast, old-field pineland fine fuel loads are predicted to become increasingly dominated by duff, hardwood and pine tree litter, and dead forb fuels. Forb-dominated herbaceous layers are generally less flammable than those dominated by grasses (Brown 1981, Platt et al. 1991). Duff and litter fuels require drier, and thus more limiting, conditions for effective fire spread (Ferguson et al. 2002). Under very dry conditions, duff accumulated over 5 or more years can potentially endanger canopy trees because of high residence times (Chapman 1947, Varner et al. 2000, Hiers et al. 2003). These predictions are consistent with measurements of fuel loads and fire behavior in old-field pineland research plots burned at 1-, 2-, and 3-y intervals at TTRS since 1959 (R.E. Masters and K.M. Robertson, TTRS, unpublished data), and shed light on the ability of native pinelands to have persisted through the potentially widely varying fire intervals occurring on the pre-settlement landscape (Platt et al. 1991).

MANAGEMENT IMPLICATIONS

This study suggests that old-field (shortleaf-loblolly) pinelands require a narrower prescription window than native (longleaf-wiregrass) pinelands to achieve maximum hardwood stem kill, thus restricting management opportunities to fewer days per year. Under average early growing-season weather conditions, old-field pinelands appear to have the best fuel conditions for burning at a 2-y fire return interval, whereas native pinelands might be burned effectively at 1–3 y following the previous burn. Pine BA tends to correspond to fire intensity within the BA range studied (1.9–19.1 m²/ha [8.3–83.2 ft²/acre]) (Grace and Platt 1995). Conversely, the results suggest that thinning to lower levels may not have a marked positive effect on total 1-h fuels (Moser and Yu 1999), although this seems counterintuitive.

The effects of fuel loading on fire behavior can be altered by varying burn prescriptions. Prescriptions need to be balanced with other management goals, including burning at the time of year appropriate for

certain wildlife species (e.g., Hermann et al. 1998, Bishop and Haas 2005) and allowing natural regeneration of pines (Lindeman et al. 1997). Managing old-field pinelands with fire for their many habitat values may require additional attention to fuel characteristics, time since burn, weather conditions, and competing management needs, in comparison to their native counterparts. Based on these results and our management experience, we recommend that old-field pineland prescriptions allow for 20–40% RH, 2 or more days since rain, and 7–10% fine fuel moisture, and that native pineland prescriptions allow 35–60% RH, 1 or more days since rain, and 10–15% fine fuel moisture.

The key variables ultimately distinguishing native from old-field pineland fuels in this study appear to be grass biomass, hardwood abundance, and pine species. Efforts to restore old-field pineland fuel structures to those found in native longleaf pine forests should focus on some combination of grass reestablishment, hardwood density control, and longleaf pine restoration (Harrington and Edwards 1999, Mulligan and Kirkman 2002). Many resources for such methods are available and have been reviewed elsewhere (Walker 1998, Kirkman and Mitchell 2002, Cox et al. 2004, Brockway et al. 2005). Restoration efforts to ameliorate fuel conditions may reduce long-term management expenses by resulting in more effective burns, reduced hardwood encroachment, and longer burn seasons. However, old-field pinelands are similar enough in fuel structure, fire behavior, and ecosystem function to provide habitat for a wide range of fire-dependent wildlife species that have become globally rare, demonstrating the feasibility of restoring valuable habitat using the relatively inexpensive tools of prescribed burning and timber thinning. In conclusion, this study provides insight into the tradeoffs between certain land-use decisions and the ability to meet management objectives using fire in southeastern U.S. pinelands.

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SPATIAL VARIATION OF FIRE EFFECTS WITHIN A *JUNIPERUS-QUERCUS* SAVANNA

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ABSTRACT

Spatial dependence among fuel, fire behavior, and fire effects has been largely overlooked in fire research of grasslands and savannas. We measured fuel loading and continuity, fire behavior, and effects of summer fire on woody plants at the Texas A&M Agricultural Experiment Station on the Edwards Plateau. Vegetation is a midgrass and shortgrass matrix interspersed by *Juniperus* and *Quercus*. Crown scorch and change in canopy cover of woody species were quantified along permanently established transects. We found that fire effects on woody plants in this savanna vary with fire behavior, constrained by variation in fine fuel load. We will use these results to test the hypothesis that a feedback between fuel load and fire effects sustains a spatially fixed vegetation mosaic across fire events in savannas. Additional research will explore interactions between spatial aspects of fuel availability and fire effects with change in fire climate.

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THE EFFECT OF THINNING AND ASPECT ON FIRE BEHAVIOR IN THE MISSOURI OZARKS OF THE CENTRAL HARDWOOD REGION

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ABSTRACT

Prescribed fire and thinning are commonly used practices in glade-savanna-woodland restoration. However, little information exists on the effects of aspect or thinning on fire behavior or fuel loading in the Central Hardwood Region. Fire behavior (rate of spread and flame length), fuel loading, and consumption data were collected in the southeastern Missouri Ozarks to determine if aspect (north- and east-facing backslopes [protected], south- and west-facing backslopes [exposed], and ridges [no aspect, <8% slope]) or treatment (thin-burn versus burn only) had an effect on fire behavior. Fuel loading differed for 1000-h fuels by aspect for pre-treatment, post-thinning, and post-thinning-burn treatments. The general pattern was a progression from exposed slopes, to ridges, to protected slopes. Thinning increased total fuel loading about 300% (from 1.53–1.93 kg/m² to 5.04–6.36 kg/m²), with 100- and 1000-h solid fuels replacing litter as the heaviest component of the total.

When burned, exposed and protected aspects exhibited greater average and maximum flame lengths and fireline intensities. Exposed aspects exhibited more intense fires compared to ridges and protected aspects. Heat per unit area and total energy release were mostly affected by treatment, not by aspect, with thin-burn treatments experiencing greater levels of both than the burn-only treatment. Comparative differences in fireline intensity, heat per unit area, and total energy release indicate that fireline intensity may be the best predictor of fire effects.

keywords: Central Hardwood Region, fire behavior, fuel loading, intensity, Missouri Ozarks, oak (*Quercus*), oak-pine (*Quercus-Pinus*).

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INTRODUCTION

Fire behavior, a product of fuels, weather, and topography, and its effects are important to foresters, fire managers, and ecologists alike for managing prescribed fire and suppression efforts. Fire managers use existing fuels and various weather parameters in formulating prescriptions to achieve certain fire behavior characteristics that meet management objectives (Johnson and Miyanishi 1995). The response of an ecosys-

tem to disturbance is dependent on the severity, frequency, and size of the disturbance itself (Romme et al. 1998). In the case of fire, severity is a function of energy released, or intensity (Byram 1959, Brown and Davis 1973, Alexander 1982, Johnson and Miyanishi 1995).

Lack of quantification of fire behavior has been a major shortcoming in many studies of fire effects. However, measures that characterize fire behavior exist that correlate well with effects; the two most common measures used to characterize fire behavior are fireline intensity (FLI) and heat per unit area (HPA) (Rothermel and Deeming 1980, Alexander 1982, Johnson and Miyanishi 1995). FLI (kW/m) is the energy released

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by the flaming front over a given time; HPA (kJ/unit area) accounts for the amount of energy released per unit of area by the flaming front (Byram 1959, Brown and Davis 1973, Alexander 1982, Johnson and Miyanishi 1995). Another measure is total energy release (TER; kJ/unit area). TER accounts for both energy released by the flaming front and energy released by consumption after the flaming front (Byram 1959, Rothermel and Deeming 1980).

FLI has been used successfully for many years as a tool to guide wildfire suppression tactics (Rothermel 1972, Albini 1976, Anderson 1982). Though success has been mixed, attempts have been made to correlate fire behavior with ecosystem response. The understanding of the effects from one fire is good to fair while that of the ecological effects of multiple fires is poor to fair, at best (Agee 1997). Overall, natural ecosystem variability combined with the variability associated with fire behavior makes correlating fire behavior with ecosystem response difficult. However, fire behavior and effects are not so unique that generalizations are impossible (Rothermel and Deeming 1980, Agee 1997).

FLI is derived using one of two methods: 1) estimated flame length and 2) combining rate of spread (ROS) with fuel consumption and fuel "low heat" of combustion (Byram 1959, Rothermel and Deeming 1980, Alexander 1982, Nelson 1986). However, both methods have flaws. Flame length and ROS observations are extremely variable through time and space (Johnson 1982) and rely mostly on human observations, which can contain bias and perception error. Additionally, fuel consumption can erroneously include energy released after the passing of the fire front (Ryan 1981, Alexander 1982). Consequently, several methods have been developed to remove a significant portion of human subjectivity from fire behavior sampling. Ryan (1981), Simard et al. (1989), Finney and Martin (1992), and Kolaks et al. (2004a) have developed or evaluated the use of fire-retardant-treated cotton string or tin-lead solder as passive flame-height sensors. Blank and Simard (1983) and Simard et al. (1989) developed an algorithm that utilizes measurements from three buried timers to derive ROS and direction of fire spread across any triangle.

TER is calculated by multiplying the weight of all fuel consumed by the energy content of that fuel minus the heat of vaporization (Byram 1959, Alexander 1982). In some cases, the total energy released can be more important than fireline intensity (Rothermel and Deeming 1980). This situation is likely to occur where heavier fuels (2.54 cm diameter or greater) ignite and burn for a period of time after the passing of the flaming front. However, TER does not account for the rate of energy release, or intensity, associated with the flaming front and cannot be considered a part of residence time or FLI (Rothermel and Deeming 1980). The effects of two fires having identical TER measurements could be completely opposite based on the proportion of energy released during the short duration of the flaming front versus the longer duration of glowing combustion.

Fire behavior information from the Central Hardwood Region is needed because fire has long played a role in the development and maintenance of oak (*Quercus*) forests. Fire, or the lack thereof, has altered vegetation structure as well as midstory and overstory composition of most oak forests in the United States. Abrams (1992) and Nuzzo (1986) contend that prairies, woodlands, and savannas have, or are becoming, dense oak forests, while oak forests, in some instances, are becoming maple (*Acer*) forests on more mesic sites. In many instances, fire is being put back into these ecosystems to restore "natural" pre-European settlement conditions or to favor oak regeneration.

Introducing fire back into hardwood landscapes may seem like the logical answer to many management issues concerning oak forests, such as oak decline or successional replacement. However, reintroduction of fire is not without consequences. Although oak is resistant to fire-caused mortality, it is still susceptible to damage that can degrade the butt log, the most valuable portion of a tree (Paulsell 1957; Scowcroft 1965; Loomis 1973, 1974, 1977; Loomis and Paananen 1989; Regelbrugge and Smith 1994). Managers, under the impression that "cool" fires will not damage trees, typically utilize "cool" backing fires when conducting "fuel reduction" burns in hardwood forests. Conversely, burns for savanna or woodland restoration are typically "hot" headfires, so as to kill or damage a portion of the overstory to reduce basal area and stocking. However, qualitative descriptors such as "hot" or "cool" are often difficult to correlate with ecologic response (Rothermel and Deeming 1980, Alexander 1982, Johnson and Miyanishi 1995). It is quite possible that backing fires may apply the same, or even more, heat to an area due to slower rates of spread and longer residence times.

Purpose

In an attempt to better quantify fire behavior under known fuel and weather conditions, we collected fire behavior data during prescribed burns in the Central Hardwood Region of Missouri. This project was part of a larger cooperative study funded by the Joint Fire Science Program. The project cooperators included the USDA Forest Service North Central Research Station, U.S. Geological Survey Northern Prairie Wildlife Research Center, University of Missouri-Columbia, and the Missouri Department of Conservation. The purposes of this study were to determine existing fuel loads and whether aspect (south- and west-facing slopes [exposed], no aspect [ridge], and north- and east-facing slopes [protected]) affected fuel loading, and fire behavior where applicable, in stands that received thinning, prescribed fire, both thinning and prescribed fire, or no management (control).

STUDY AREA

The study area was located in the southeastern Missouri Ozarks near Ellington, Missouri (lat 37.242°N, long 90.969°W), on land managed by the

Missouri Department of Conservation (Figure 1). In an effort to minimize variation caused by potential vegetation differences, study sites were all located within the Black River Oak/Pine Woodland/Forest Hills Landtype Association (Black River Hills LTA) per the Missouri Ecological Classification System (Meinert et al. 1997, Nigh and Shroeder 2002). The Black River Hills LTA, characterized by strongly rolling to hilly lands with steep slopes and flat land found only in creek and river bottoms, historically comprised oak and oak–pine (*Quercus–Pinus*) woodlands and forests. These forest types still dominated but were second growth and had grown more closed due to fire suppression (Nigh and Shroeder 2002). The Black River Hills LTA was in the center of one of the largest blocks of forest in the Midwest that also supports a substantial timber industry (Nigh et al. 2000).

METHODS

Site Selection

Stands selected for the study had no known forest management activities or wildfire for at least 30 y. All stands were fully stocked (average basal area of 9.75 m² [105 ft²] and 874 trees/ha [354 trees/acre]) according to Gingrich (1967) (average 92%) and composed primarily of oak–hickory (*Quercus–Carya*) and oak–pine forest types. Treatments were replicated across 3 complete blocks. Each block had 12 stands (3 aspect classes and 4 treatments), with each stand being an aspect–treatment unit. Stand area ranged from 3 to 6 ha (12–15 acres). Aspect classes included exposed backslopes (135–315°), ridge, and protected backslopes (315–135°) (Nigh et al. 2000).

Treatments

Treatments were randomly assigned on the study sites and included commercial thinning, prescribed burning, both commercial thinning and prescribed burning, and no treatment (control). Commercial thinning of the overstory occurred during the summer and early fall 2002. Leave-tree preference was given to individuals having relative fire tolerance, good form, and canopy dominance. A mark–leave method was utilized to select leave trees. Merchantable unmarked trees were felled and logs skidded to landings for processing prior to removal from the site. Tops were left in place along with cull sections of logs. Any remaining unmarked trees with diameter at breast height (dbh) >3.5 cm were cut after the harvest was complete. In addition to mechanical harvest, windthrow aided in reducing stocking to 41% (average basal area of 3.9 m² [50 ft²] and 217 trees/ha [155 trees/acre]) according to the Gingrich (1967) stocking chart. The target stocking level was 60% because it is commonly used in intermediate cuttings, shelterwood systems, and savanna–woodland restoration (Johnson et al. 2002).

Prescribed burns were conducted during spring 2003, using prescription conditions common in the

Missouri Ozarks (Table 1). Prescribed burns were ignited with a ring-fire method while simultaneously igniting the ridges. This firing method was used because it is the most common firing technique in the region. The objectives of the prescription were to reduce fuels and kill woody stems <3.81 cm dbh.

Data Collection

Fuel Loading

We inventoried fuels using a modified transect intercept method with a transect emanating in a random direction from 15 random points within each stand. Pre-thinning data were collected in winter 2002. Post-thinning and pre-burn data were collected during winter 2003. Post-burn sampling was completed after the burns in spring 2003 (Kolaks et al. 2003, 2004b). Woody fuels were separated into four size classes: 0.0–0.64 cm (1-h), 0.64–2.5 cm (10-h), 2.5–7.6 cm (100-h), and >7.6 cm (1000-h). The 1000-h fuels were further separated into *rotten* and *solid* categories. From each sample point, 1- and 10-h fuels were inventoried along a 1.8-m segment, 100-h fuels along a 3.7-m segment, and 1000-h fuels along the entire 15.2-m length of the transect. Fuel height and litter and duff depths were measured at 1.5-m intervals along the fuel transect, starting 0.3 m from the origin (Brown 1974, Brown et al. 1982, Grabner 1996, National Park Service 2001). Litter and herbaceous samples were collected from a 0.2-m² (2-ft²) clip-plot located at the end of each fuel transect. Samples were then dried at 60°C to a constant weight and reported on a dry-weight basis (Grabner 1996).

Fuel Consumption

Fuel consumption was calculated for each time-lag class at the transect level. Post-burn fuel loading was subtracted from pre-burn fuel loading (Kolaks et al. 2004b). In rare instances, fuel loading in given time-lag class increased because of incomplete combustion of fuels in the next largest time-lag class (e.g., a 100-h fuel partially consumes to the size of a 10-h fuel). For the analysis, negative consumption values were ignored.

Rate of Spread, Flame Height, and Flame Tilt Angle

Modified ROS clocks and passive flame-height sensors were used to collect ROS data and flame-height data from three randomly chosen points out of the 15 preexisting points within each stand. In order to objectively determine flame length, two measures were recorded at each point. First, flame height was recorded by an array of passive flame-height sensors calibrated for the Central Hardwood Region (Kolaks et al. 2004a). Each array comprised 12 strands of fire-retardant–treated cotton string suspended between two wires, one at fuel bed height and the other approximately 2.3 m above the fuel bed. Second, trained observers used visual aids (the clinometer in a Silva Ranger® compass and a clear sheet of plastic equipped

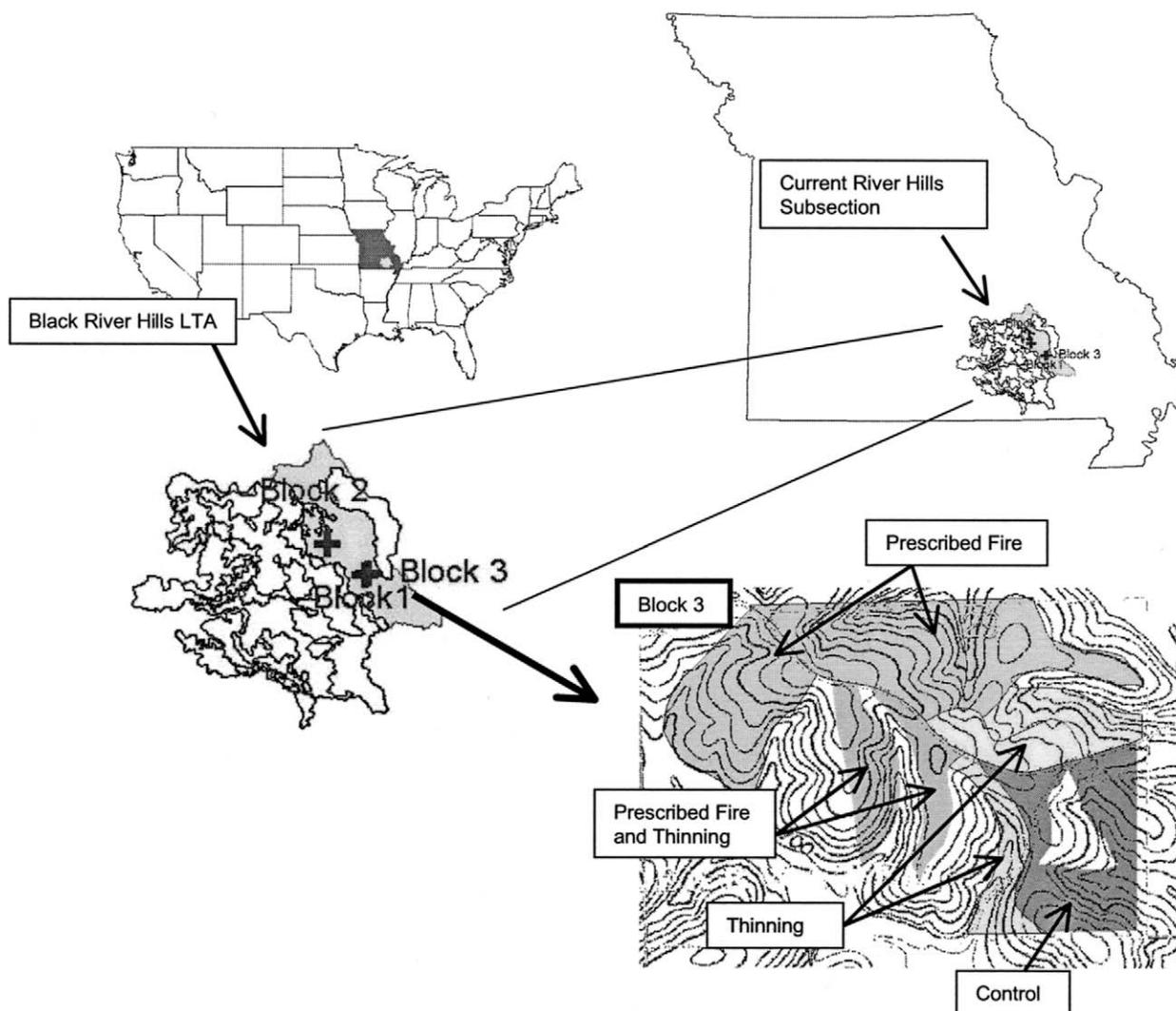


Fig. 1. Location of study sites and arrangements of treatments (block 3 used as an example) in forest stands treated with prescribed fire or with both thinning and prescribed fire, Central Hardwood Region, Missouri Ozarks, 2002–2003.

with a bubble level and marked with various angles from 0 to 90°) to determine flame-tilt angle as the fire front passed through the flame-height sensor arrays. Flame-tilt angle, which is combined with flame height to derive flame length, can be measured at a greater

Table 1. Weather parameters and fuel moisture prescription used for prescribed burning and observed values in forest stands treated with prescribed fire or both thinning and prescribed fire, Central Hardwood Region, Missouri Ozarks, 2002–2003.

Parameter	Prescription	Observed	
		Average	Range
Temperature (°C)	7–18	18	7–28
Mid-flame wind (m/s)	0–3.1	1.1	0–3.4
Relative humidity (%)	25–45	22.4	9–46
Percent fuel moisture			
1-h	5–10	5.4	5–6
10-h	8–15	9.8	8–12
100-h	12–18	13.7	13–18
1,000-h	>20	17.6	17–20

distance by fewer observers than direct observation of flame length (Ryan 1981).

Average flame lengths (AFLs) were then derived by averaging flame heights indicated by sensors in each array (Kolaks et al. 2004a) and applying the corresponding flame-tilt angle (Ryan 1981). Estimated maximum flame length (MFL) was derived using the tallest flame height recorded by a sensor array and applying the flame-tilt angle. These methods eliminated a large component of variation imposed via human observations by removing subjectivity and perception discrepancies.

To measure ROS, five clocks were located at each point, one at the point and the others 15.24 m in each cardinal or sub-cardinal (NE, SW, etc.) direction, depending on the locations of fire-spread obstructions such as logs and rock outcroppings. ROS and direction were calculated using at least three measurements from the buried clocks (Simard et al. 1984). Unfortunately, only a fraction of the ROS clocks worked properly. Over two-thirds of the clocks endured cold temperatures and precipitation for several days after the burn

due to circumstances beyond our control. Clocks recovered immediately after the burn worked properly and provided good data. However, sufficient data were not available to conduct statistical analysis. Given that the FLI can be determined by two methods, and that the two methods are correlated with each other (Byram 1959, Nelson 1986), estimates of ROS were derived by dividing FLI by energy released by 1-h woody and litter fuels (Equation 1). We assumed that only litter and 1-h woody fuels were consumed by flaming front. Given that $I = HWR$ (Byram 1959), then

$$R = I/HW \quad (1)$$

where R = rate of spread (m/s), I = FLI (kW/m), H = low heat of combustion (18,577 kJ/kg [adjusted for heat of vaporization]), and W = weight of fuel consumed (1-h woody and litter [kg]).

Environmental Measurements

Weather parameters, including eye-level wind speed and direction, 10-h fuel moisture and temperature, relative humidity (RH), and air temperature, were measured and recorded every 15 min with a Campbell® Scientific automated weather station (Campbell Scientific, Logan, UT). The station was placed on a ridge upwind and adjacent to the sites under the leafless canopy. One-, 100-, and 1000-h fuel moistures were calculated using data from two automated weather stations that experienced similar weather patterns located in relatively close proximity (15 and 24 km) to the study sites.

Fireline Intensity, Heat Per Unit Area, and Total Energy Release

FLI and HPA were calculated using average flame lengths for each of the fire behavior plots. Byram's (1959) FLI equation was used (Equation 2):

$$FLI = 258F_L^{2.17}, \quad (2)$$

where FLI = fireline intensity and F_L = flame length (m). HPA (Equation 3 [Rothermel and Deeming 1980]) was calculated using the estimated ROS (Equation 1) for each of the fire behavior points. Using average flame length alleviates issues associated with consumption of fuels after the passing of the flaming front.

$$H_A = (60I)/R, \quad (3)$$

where H_A = heat per unit area (kJ/m²), I = FLI (kW/m), and R = rate of spread (m/min).

To obtain TER, post-burn fuel loading was subtracted from pre-burn fuel loading (Kolaks et al. 2004b). Using methods outlined by Byram (1959) and Brown and Davis (1973), TER was derived for each time-lag class (Equation 4):

$$\begin{aligned} \text{TER} = & \text{weight of fuel consumed} \\ & \times (\text{heat of combustion for oak} \\ & - \text{heat of vaporization}), \end{aligned} \quad (4)$$

where heat of combustion for oak = 19,215 kJ/oven-

dry kg, and heat of vaporization = 24 kJ × (% fuel moisture) + 573 kJ. Fuel moisture was assumed to be applicable over the entire site for a given day. TER was then grouped into 1-h fuel loading (litter and 1-h woody fuels), all fuel, and the percentage of total TER that was released by 1-h fuels. A reduction for radiation, suggested by the original methodology (Byram 1959), was not made because there is no sound basis available for estimating radiant heat as a proportion of the total energy output, and heat "losses" from radiation actually contribute greatly to fire behavior (Alexander 1982).

(All intensities are reported in kilojoules despite British thermal units [Btu] being the most common output used by management models for FLI. However, FLI can be easily converted from kW/m to Btu/ft per second by dividing by 3.4592 [Alexander 1982]. HPA, TER, and 1-h TER can be divided by 1.0551 to derive Btu/ft².)

Data Analysis

Analysis of variance was used to determine if differences in fuel loading and fire behavior differed due to treatment and aspect. Data were analyzed using the MIXED procedure in SAS (SAS Institute, Cary, NC). This procedure was used because it allows covariates to vary within a subject (Wolfinger and Chang 1995). We used $\alpha \leq 0.05$ to test differences among aspects, treatment, and aspect-treatment interactions in TER. Because we did not include weather factors or slope as explanatory variables, and because of fire behavior's variable nature, $\alpha \leq 0.10$ was used for the same differences in AFL, MFL, and HPA.

RESULTS AND DISCUSSION

Although data may appear conclusive, it is important to remember that, due to the variable nature of fire behavior, the results of this study are only representative of the given fuel conditions, weather, and topographic conditions. Variables not taken into account during this analysis, such as possible differences in slope of the protected and exposed backslopes, may account for some variation. Dormant-season "canopy covers" can also differ between stands, allowing for differing solar insulation between stands within an aspect class. Burning under any other conditions, even the same fuel conditions with different weather conditions, may yield surprisingly different results. Furthermore, the estimation of ROS from the back-calculation of an empirical formula should cast a degree of uncertainty on HPA.

For the most part, all burns were conducted within prescription. Average RH was below prescription, but ignition operations were completed before the RH dropped below the lower threshold (Table 1). Although prevailing wind direction for the Missouri Ozarks is south or southwest, winds were primarily out of the north-northwest during the burning period. Because exposed slopes are typically drier (Nigh et al. 2000), differences in all fire behavior parameters, especially

between exposed and protected backslopes, may have been more pronounced had there been a southerly wind.

Fuel Loading and Consumption

Data for fuel loading and consumption were presented in Kolaks et al. (2004b). In unthinned stands, total fuel loading varied from 1.53 to 1.93 kg/m² (Kolaks et al. 2004b). Significantly different loading due to aspect was only present in 1000-h solid fuels. All other time-lag classes, including fine fuels (litter and 1-h woody), did not significantly vary by aspect (Kolaks et al. 2003).

In thinned stands, total fuel loading ranged from 5.04 to 6.36 kg/m² (Kolaks et al. 2004b). Significant differences did not exist in fine fuels between aspects. However, significant differences existed in the 10-h time-lag class between exposed backslopes and ridges, and the 100-h time-lag class between exposed slopes and both ridges and protected slopes. On average, fuel height was 26.7 cm greater while litter depth was 2.5 cm lower following thinning (Kolaks et al. 2004b).

Average and Maximum Flame Length

As a whole, aspect significantly impacted AFL. On average, AFL was greater on the backslopes than on ridges, with exposed backslopes having significantly greater flame lengths than ridges and protected backslopes ($P = 0.010$ and 0.083 , respectively) (Table 2). AFL was also significantly different among aspects within the thin–burn treatment, with exposed slopes being significantly greater than ridges and protected slopes ($P = 0.012$ and 0.064 , respectively). However, despite similar differences between aspects in the burn-only treatment, high variability masked any potential difference. On average, AFL was greater for the thin–burn treatment compared to the burn-only treatment, albeit not significant.

Estimated MFL behaved similarly to AFL. Overall, MFL was greater on backslopes than on ridges, with exposed backslopes having significantly greater flame lengths than ridges and protected slopes ($P = 0.006$ and 0.082 , respectively). MFL was also significantly different among aspects within the thin–burn treatment, with exposed slopes being significantly greater than ridges and protected slopes ($P = 0.005$ and 0.025 , respectively) while not being significantly different among any aspect class in the burn-only treatment. Significant treatment differences within aspect class were not detected except for exposed slopes, with the thin–burn treatment having significantly greater MFL than the burn-only treatment ($P = 0.038$) (Table 2).

Despite exposed slopes having significantly less fuel loading in the 100-h time-lag category, significantly greater average and maximum flame lengths occurred on exposed backslopes than on ridges or protected backslopes. This could be attributed to drier fuels due to solar exposure. Again, a southerly wind may have created a more pronounced difference between exposed and protected slopes.

Table 2. Fire behavior characteristics by aspect and treatment in forest stands treated with prescribed fire or both thinning and prescribed fire, Central Hardwood Region, Missouri Ozarks, 2002–2003. Different letters indicate significant difference within a row (lowercase) or column (uppercase).

Treatment	Aspect			Treatment average
	Exposed	Ridge	Protected	
Average flame length (m)				
Burn	0.54aA	0.30aA	0.45aA	0.44A
Thin–burn	0.86aA	0.35bA	0.52bA	0.58A
Aspect average	0.72a	0.33b	0.49b	
Estimated maximum flame height (m)				
Burn	1.73aA	0.90aA	1.62aA	1.42A
Burn–thin	2.98aB	1.1bA	1.55bA	1.89A
Aspect average	2.36a	1.00b	1.61b	
Byram's fireline intensity (kW/m)				
Burn	100aA	21a	62a	62A
Burn–thin	242aB	42b	118b	131A
Aspect average	170a	35b	26b	
Heat per unit area (kW/m ²)				
Burn	1,166A	1,196A	1,011aA	1,136A
Burn–thin	1,227bA	1,330A	1,448acB	1,335B
Aspect average	1,197	1,263a	1,262a	
Total energy release (TER) (kJ/m ²)				
Burn	2,592aA	2,579aA	2,485aA	2,552A
Burn–thin	3,542aB	3,610aB	3,788aB	3,656B
Aspect average	3,066a	3,136a	3,110a	
1-h TER (kJ/m ²)				
Burn	2,251aA	2,273aA	2,145aA	2,222A
Burn–thin	2,302aA	2,386aA	2,486aB	2,391B
Aspect average	2,276a	2,330a	2,317a	
TER accounted for by 1-h fuels (%)				
Burn	80aA	85aA	84aA	83A
Burn–thin	62aA	77aA	68aA	69B
Aspect average	71a	81a	76a	

Average and maximum flame lengths in thinned stands did not accurately reflect the average across the entire stand. Passive flame-height sensors could not be installed through logging slash. Rather, AFLs more accurately depict the areas between slash piles as the thinning influenced them. The trained observers visually estimated flame lengths off of slash. Estimates included average and maximum flame lengths. In general, flame lengths from slash averaged 4.3 m, with MFLs of 15.2 m being common.

Fireline Intensity

Because FLI was derived from flame length, FLI behaved almost identical to flame length. Treatments (excluding slash flame-length data) did not significantly affect FLI, despite a minor difference of 19 kW/m (Table 2). On average, overall FLI was greater on backslopes than on ridges; exposed backslopes had a significantly greater FLI than ridges and protected slopes ($P = 0.025$). FLI was also significantly greater by aspect in the thin–burn treatment, with exposed slopes having significantly greater intensities than ridges and protected slopes ($P = 0.017$ and 0.100 , respectively), while not being significantly different from any aspect class in the burn-only treatment. Despite the thin–burn treatment having greater FLI than

the burn-only treatments by a magnitude of 2 on average, treatments were significantly different only on exposed slopes ($P = 0.025$) (Table 2).

Heat Per Unit Area

HPA was only significantly affected by aspect in the thin-burn treatment. Significant differences within an aspect class due to treatment only occurred on protected slopes. Overall, the thin-burn treatment had a greater HPA than the burn-only treatment (Table 2). Differences are difficult to attribute because ROS, a divisor in calculating HPA, was calculated from FLI. The overall difference between treatments can be attributed to obviously greater fuel loads in thinned stands and could be greater than indicated because average flame length, used in the calculation of FLI, did not include the flame lengths off of slash. The difference in HPA on protected slopes between treatments is most likely the result of greater contributions of fuel from thinning on protected slopes compared to exposed slopes (Kolaks et al. 2004b).

Although significant differences existed between aspect classes in the thin-burn treatment ($P = 0.017$), the overall difference between aspects did not exceed 221 kW/m². Since HPA ranged from 1,089 to 1,441 kW/m², the significance of a 221-kW/m² difference on ecological response is unknown. However, on protected slopes, we saw a change of nearly 402 kW/m² difference between treatments ($P = 0.009$). Differences in HPA between slopes may be the result of weather conditions (i.e., north wind) or time of the burning period (i.e., drier portion of the day). A difference between treatments could be attributed to greater fuel loading in thinned stands where the thin-burn treatment received almost one-third more heat than the burn-only treatment on protected backslopes. Though the importance of this contrast is unknown, a difference in ecological response would not be surprising on protected backslopes between treatments.

Within treatments, the ridges received almost identical HPA as the backslopes. The lack of difference in HPA between backslopes and ridges (Table 2), despite the significant difference between the backslopes and ridges in AFL and FLI (Table 2), indicate that a backing or slower-moving fire may not actually be a "cool" fire.

Total Energy Release

TER was significantly higher in the thin-burn treatment compared to the burn-only treatment in every aspect class and overall by treatment (Table 2), which can be attributed to the greater availability of fuels after thinning. There were no significant or nearly significant differences as a result of aspect, despite differences in flame length and FLI. The lack of difference due to aspect indicates that FLI or flame length may not have a bearing on consumption.

One-hour TER responded to treatment similarly, but to a lesser degree than TER, likely a result of the minor contribution of 10-h time-lag fuels in the combustion process. Overall, the thin-burn treatment was

significantly greater than the burn-only treatment ($P = 0.001$). Within aspect class, the thin-burn treatment was significantly greater than the burn-only treatment on protected aspect class only ($P = 0.033$) (Table 2).

The percentage of TER accounted for by 1-h TER was significantly different overall by treatment ($P = 0.022$), with 1-h fuels (litter and 1-h woody) being responsible for an average of 83% of TER in the burn-only treatment and 69% in the thin-burn treatment (Table 2). One-hour fuels accounted for 38–52% of total fuel loading in unthinned stands and 13–15% of total fuel loading in thinned stands (Kolaks et al. 2003, 2004b). This suggests that, despite large amounts of heavy fuel (100-h and greater), 1-h fuels have the greatest influence on fire behavior and energy release (Byram 1959, Davis 1959) under typical prescribed burning conditions (Table 1). Brown (1972) found similar results with fuel <1 cm (0.39 inch) in diameter accounting for <30% of the total loading and half of the total weight loss in ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) slash fires. Furthermore, for slash and other fuels containing a similar mixture of particle sizes, particles <2.54 cm (1 inch) in diameter essentially provide the energy that characterizes propagation of the flaming front.

Despite fine fuels having the greatest impact on characteristics of fire behavior, the proportion of energy released by heavy fuels may be just as, or even more, important (Rothermel and Deeming 1980). Anderson (1969) found that residence time in minutes was equal to 3.15 times the particle diameter in centimeters. Increased consumption of larger fuels, as was the case in the thin-burn treatment, would indicate longer residence times from either open flame or glowing combustion (Kolaks et al. 2004b).

MANAGEMENT IMPLICATIONS

When relating FLI, HPA, and possibly TER to fire effects on components of an ecosystem, such as trees, it is important to consider that components of the ecosystem may be more or less resistant to different amounts of heat for different periods of time. It is also important to note that this resistance may change according to age, season, or other environmental conditions such as drought (Whelan 1995). Past research in oak-hickory forests of the Central Hardwood Region indicate that top-kill, mortality, and damage are highly correlated and directly related to the height of stem-bark char and inversely related to diameter at breast height (Paulsell 1957; Scowcroft 1965; Loomis 1973, 1974; Loomis and Paananen 1989; Regelbrugge and Smith 1994). Height of stem-bark char can be correlated with flame height and length (Cain 1984), and subsequently correlated with FLI (Byram 1959, Alexander 1982, Nelson 1986). Given that higher stem-bark char would indicate greater FLI, these variables suggest that oak-hickory forest trees may be more resistant to low heat for longer periods of time (low-intensity fires), as opposed to high levels of heat for shorter periods of time (high-intensity fires), even

though both situations may be indicative of near-identical HPA and TER. Further, this suggests that FLI may be a better predictive variable for fire effects than HPA or TER.

Other woody and herbaceous plants may respond similarly or differently depending their methods or pathways of "resistance." Future study of these responses will not only need to account for fire behavior measures independent of each other, but also the response to varying combinations of fire behavior measures (i.e., high FLI versus low HPA). These measures combined with the biologic and physical environment of a burn can aid greatly in the explanation of fire effects.

CONCLUSION

When utilizing a ring-headfire-ignition method, differences among aspects in flame length and FLI may have significant impact on planning, suppression efforts, and ecological response, despite HPA and TER not varying greatly. Increased flame length and FLI on backslopes indicate that control lines for prescribed and wildland fires should be placed on ridges, not on slopes, especially where fire will impinge from below. Also, given that flame lengths of slash piles averaged 4.2 m (14 ft), with a maximum of 15.2 m (50 ft), control lines should be located a sufficient distance away from such fuels to reduce the chance of spot fires.

HPA and TER were affected most by treatment, with little effect due to aspect. It is possible that a backing fire could have greater impact than a headfire given longer residence times for backing fires. Managers, ecologists, and researchers alike should evaluate these discrepancies, especially in HPA versus FLI, between different fire behaviors (i.e., head and backing) caused by ignition techniques. We think FLI may be the best predictor of fire effects within homogeneous fuel conditions, given that it varied due to slope, whereas HPA and TER did not. Also, because litter and 1-h woody fuels (1-h fuels) account for the greatest proportion of energy released during a fire, they will have the greatest effect on FLI and HPA, despite total fuel loading.

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TESTING MUTCH'S HYPOTHESIS IN SOUTHEAST QUEENSLAND: PLANT FLAMMABILITY REVISITED

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ABSTRACT

Mutch's Hypothesis states that some plants have evolved traits that enhance their own flammability in order to increase the probabilities of catastrophic wildfires, in order to eliminate competitors. In addition, these plants have also needed to evolve characteristics that would allow the plants to regenerate rapidly after these fires. Other studies have attempted to examine Mutch's Hypothesis, and recently some models have shown that flammability traits can be genetically inherited.

This study investigated the flammability characteristics of foliage and litter collected from subtropical rainforest (considered nonflammable) and sclerophyll forests (considered highly flammable) in southeast Queensland, Australia. Flammability was calculated using a specialized combustion chamber that accurately simulated conditions comparable to wildfires experienced in Australia. Flammability was assessed using predetermined characteristics such as flame residence time, consumption rate, rate of spread, and ignition potential. Heat release rate (or "sustainability") was shown to be an accurate indicator of flammability. Litter accumulation and decomposition rates in both vegetation communities were also assessed. Additional experiments involving the exchange of litter between forest types were also conducted to examine the effect of vegetation community on decomposition rates. Preliminary analysis of the results indicates a division in the flammability characteristics between flammable and nonflammable plant communities. This is also demonstrated in the litter experiments, in which the results indicate similar energy contents but differing rates of decay and accumulation. Mutch's Hypothesis may be able to explain the differences in flammability of vegetation communities that have the same climatic and meteorological conditions.

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FUEL BED CHARACTERISTICS AND FIRE BEHAVIOR IN CATBRIER SHRUBLANDS

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ABSTRACT

Invasion of grasslands by woody shrubs can alter existing fire regimes and give rise to problem fire behavior. Invaded areas are likely to burn less often but with more intensity. Abandoned pastures on the Elizabeth Islands, Massachusetts, that have been invaded by the woody vine catbrier (*Smilax rotundifolia*) follow this pattern. We evaluated the usefulness of standard and custom fuel models (CFMs) for predicting fire behavior in a 0.2-ha (0.5-acre) experimental burn. Developing the custom fuel model required characterizing fuel load and fuel bed depth of catbrier fuels, which contain a dense mat of vines with nearly 100% cover and a height of 1 to 2 m. This was done by measuring the height of litter and shrub components of the fuel bed; estimating cover by point-intercept sampling; and harvesting live vines and leaves, woody and non-woody litter, and dead vines from 1-m² plots. From these data, we developed regression equations to estimate fuel load using fuel bed depth.

Measured 1-h dead fuel loads (23 t/ha) were greater than that of any standard fuel model. Total (live and dead) 1-h fuel loads were accurately predicted by shrub height ($R^2 = 0.82$). All standard fuel models (SFMs), including SFM 4 (chaparral), underestimated flame length observed during an experimental burn conducted in mid-June 2004 following leaf-out, while our custom fuel model more accurately predicted these values. Observed flame length was 5 m (17 ft) (CFM prediction 4.9 m [16 ft]) and rate of spread 11 m/min (37 ft/min) (CFM prediction 14 m/min [46 ft/min]). Results of our work will aid in developing fuel reduction programs for managers interested in restoring early successional habitats, providing guidance for suppressing wildfires, and implementing prescribed fire management in catbrier-dominated habitats.

keywords: Atlantic Coastal Plain, BEHAVE, catbrier, fire intensity, fuel loads, fuel models, Massachusetts, shrublands, *Smilax rotundifolia*.

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INTRODUCTION

Catbrier (*Smilax rotundifolia*) is a thorny vine that invades abandoned pastures, grasslands, and open woodlands of the Atlantic Coastal Plain. On the Elizabeth Island chain southwest of Woods Hole, Massachusetts, catbrier has invaded large expanses of maritime grasslands, forming dense thickets of near 100% cover (Richburg 2005). These thickets reduced native plant species diversity, degraded the recreational value of the land, and altered fire regimes (Schroeder 2002, Richburg 2005; T. Simmons, Massachusetts Division of Fisheries and Wildlife, personal communication) on the islands. Shrubs and woody vines such as catbrier have several characteristics that can produce extreme fire behavior and alter fire regimes. These include their often having a higher volatile chemical content that makes them more flammable than many other wildland fuels, a high percentage of dead stems that require less heat to ignite, and a ratio of fuel to air (packing ratio) that is nearly ideal for promoting fire spread (Miller 1994). An example is the invasion of South African grasslands by woody plants, which reduced fine fuels in the understory, resulting in a decrease in fire frequency. When the fuels in shrub crowns did ignite,

fire behavior was more intense than in uninvaded areas (van Wilgen and Richardson 1985).

Resource managers in the Northeast use computer-generated fire behavior models to aid in planning for prescribed fire and wildfire control, and in evaluating the potential effectiveness of fuel management options. The most common fire behavior software is the BEHAVE Fire Behavior Prediction and Fuel Modeling System, which utilizes user-defined fuel, weather, and topographic inputs to predict fire behavior (Andrews and Bevins 1999). BEHAVE predictions are driven by mathematical algorithms that accurately predict fire behavior characteristics in many fuel types (Rothermel 1972). BEHAVE can be used either with standard fuel models (Anderson 1982) or with custom fuel models developed from parameters entered by the user.

Fuel properties that influence fire behavior include fuel loading, fuel size-class distribution, surface-area-to-volume ratio, packing ratio, fuel continuity, and fuel bed depth. These properties, along with heat content and live fuel characteristics, are the most important determinants of fire behavior (Miller 1994). Sampling of shrub fuel beds has often focused on fuel size and loading. To simplify collection of these data, basal diameters, crown diameters, and shrub heights have been used as indices of the total weight of stems (or of specific components, e.g., leaves, stems of a given size

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class, etc.) (Telfer 1969, Brown 1976, Schlessinger and Gill 1978, Gray and Schlesinger 1981, Brown et al. 1982). To relate easily measured plant characteristics to weight of common desert plants, Ludwig et al. (1975) measured canopy height, diameter, and shape to calculate canopy volume. Plants were then harvested and canopy volume was correlated with oven-dried weight ($R^2 > 0.90$) for many plants. Rittenhouse and Sneva (1977) developed predictive equations ($R^2 > 0.90$) for aboveground shrub weight by measuring crown width, crown area, and total plant height.

Although we know, anecdotally, that fires burn intensely in catbrier, there has been no attempt to document how this species alters fuel beds or to quantify fire behavior in thickets where it dominates. The goal of our research was to quantify the fuel characteristics and fire behavior that result from catbrier invasion in a coastal grassland. We describe the fuel bed of catbrier thickets using direct measurements and then use that information to develop and test a custom fuel model to predict its fire behavior.

STUDY AREA

The study area consisted of catbrier thickets in the Protected Field area of Naushon Island, Massachusetts. The 37.5 ha of the Protected Field were historically maintained as grassland by sheep and, to a lesser extent, cattle grazing before agricultural activities were largely abandoned in the early 20th century (Schroeder 2002). In its current condition, the Protected Field has large, nearly impenetrable monocultures of catbrier interrupted by patches of grasses and sedges (chiefly Pennsylvania sedge [*Carex pensylvanica*]), black huckleberry (*Gaylussacia baccata*), and the exotic Scotch broom (*Cytisus scoparius*).

Catbrier is found in 33 U.S. states and in all counties in Massachusetts (Carey 1994, Sorrie and Somers 1999). Also known as greenbrier and roundleaf greenbrier, this native woody vine with long thorns along the entire length of its stem uses tendrils to climb to heights of 3–6 m in invaded woodlands and can spread over shrubs and herbaceous plants in open areas. It regenerates vegetatively from rhizomes (sprouting prolifically following fire [Richburg 2005]) and can form dense thickets with up to 20,000 stems/ha (48,000 stems/acre) (Morong 1894, Niering and Goodwin 1962, Carey 1994).

METHODS

Describing and Quantifying the Catbrier Fuel Bed

To determine catbrier fuel bed characteristics (fuel depth and loading, cover, vertical fuel continuity, fuel particle surface-area-to-volume ratio, and packing ratio), we sampled 9 quadrats during the summer of 2003. We used a stratified random sampling design of low (<60 cm), medium (60–120 cm), and high (>120 cm) canopy heights to capture the range of fuel loads (Ludwig et al. 1975). Quadrats 1 m² in area were randomly located within each height class and sampled

for percent cover of catbrier and fuel depth (litter and shrub), using point-intercept sampling (10 points/quadrat; Mueller-Dombois and Ellenburg 1974). Vertical continuity (3-dimensional cover) of the fuels was recorded by noting the heights and types of fuels (alive or dead, stem or leaf) intersecting each point throughout the fuel column (Ohman 2006). All live and dead vegetation and litter was then harvested from each 1-m² quadrat, collected, and taken to the laboratory for further processing. After stems were harvested, stem density was tallied by counting the stumps in the quadrat.

In the laboratory, 20 sections of catbrier, randomly selected from among the nine sample plots, were measured for diameter to determine fuel class size and surface-area-to-volume ratio. All of the material was then sorted into live stem, live leaf, and dead stem components. Litter samples were sorted into herbaceous (leaf and grass-sedge) and downed woody fuels. Sorting ratio was determined by entering component fuel loads and depths into the TSTMDL (Test Model) module of BEHAVEW (DOS-based version).

Predicting Catbrier Fuel Load

Models to predict the weight of catbrier were developed separately using three fuel characteristics: vertical continuity (3-dimensional cover), stem density, and shrub height. The strength of predictors was determined using linear regression to calculate coefficients of determination (R^2).

Custom Fuel Model Development and Evaluation

We modeled fire behavior using BEHAVEW and BEHAVEPLUS3 software. Inputs to the model included fuel load and depth, surface-area-to-volume ratio, and heat content by fuel size class and category (litter, grass, slash, and shrubs). A preliminary model was developed using, for the most part, data we collected on Naushon Island. Standard estimates were used for parameters that we were unable to measure directly.

A prescribed fire was conducted on 13 June 2004 on a 0.2-ha (0.5-acre) research plot in the Protected Field. Eight samples each of litter, live leaves, and live stems were gathered from throughout the plot and returned to the laboratory to determine fuel moisture (Ohman 2006). Dead stem fuel moisture was measured in the field using a protimeter capable of recording fuel moistures >7%. Key weather parameters were measured before, during, and after the burn, including state of the weather (an estimate of the degree of cloud cover and precipitation status), ambient air temperature, relative humidity, and wind speed and direction. Wind speed was measured at 1.5 and 2 m (5 and 7 ft) above the ground (i.e., at approximately midflame height) using a digital anemometer.

The fire was ignited as a headfire with a drip torch and allowed to burn freely without the influence of backing or flanking fires. Flame lengths and rates of spread were compared to 1.2-m (4-ft) iron poles with horizontal arms at 0.3-m (1-ft) intervals placed at

Table 1. Average fuel bed characteristics (with 95% confidence intervals [CI]) in a catbrier-dominated grassland, Naushon Island, Massachusetts, 2003.

Characteristic	n	Mean	95% CI	
			Lower	Upper
Shrub height (m)	9 plots	1.00	0.74	1.26
Percent cover	9 plots	99	96	100
Total fuel load (t/ha)	9 plots	22.8	16.4	29.4
Basal stem diameter (cm)	20 stems	0.56	0.52	0.59
Percent dead	9 plots	72	66	79
Relative packing ratio		0.79		
Heat content (kJ/kg)		18,622		

6.1-m (20-ft) intervals within and parallel to the expected path of the headfire. We recorded the average length of flames as they reached the pole and the time it took the headfire to travel the 6.1-m sections between the poles. The entire burn was video-recorded to provide verification of field measurements taken during the burn.

The custom fuel model was evaluated by comparing BEHAVEPLUS3 outputs (predicted) for flame length and rate of spread—using as environmental inputs the fuel moisture and weather conditions at the time of the burn—with observed fire behavior.

Sensitivity Analysis

We performed an analysis to evaluate the sensitivity of our custom fuel model to variations in the following parameters: 1-h dead fuel load, live fuel load, heat content, fuel bed depth, surface-area-to-volume ratio, and moisture of extinction. Using environmental variables from the June 2004 burn, each of these parameters was increased and decreased by 5, 10, 20, and 40%, while holding other input parameters constant. Because changing the heat content by 40% produced an input value outside of the bounds acceptable to BEHAVEPLUS3, heat contents were changed by ± 5 , 10, and 20%, and to the upper (27,933 kJ/kg) and lower (13,966 kJ/kg) acceptable limits. For each run, the resulting flame length and rate of spread were recorded. We then compared the ratio of degree of change in the output to change in the input (Dell'Orfano 1996).

RESULTS

Describing and Quantifying Catbrier Fuel Beds

Fuel bed characteristics are summarized in Table 1. The average quadrat sampled was 1.0 m (3.3 ft) tall, had 99% cover, and was vertically continuous (as evidenced by the presence of 1-h fuels throughout the column). The total fuel load was composed (by weight) of 37% litter, 35% dead stems, 21% live stems, and 7% live leaves. The average diameter of catbrier stems 5 cm (2 inches) above their base was 0.56 cm (0.22 inches), with none >0.64 cm (0.25 inches); thus, the entire fuel bed was composed of 1-h (fine) fuels.

We calculated the surface-area-to-volume ratio for the catbrier fuel model by combining directly mea-

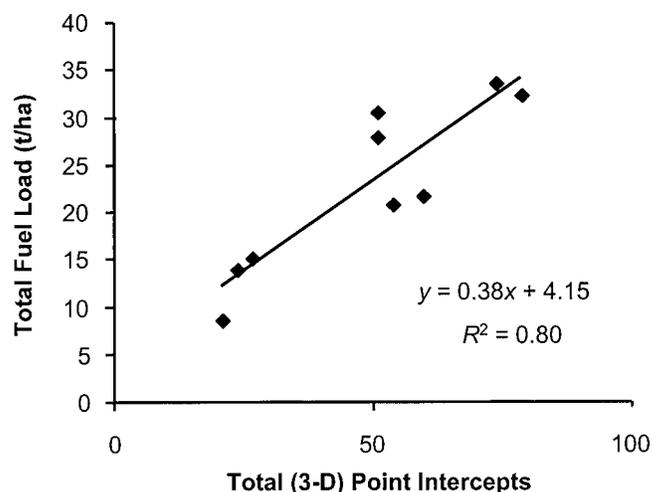


Fig. 1. Three-dimensional (3-D) cover as a predictor of total fuel load in catbrier fuel beds on Naushon Island, Massachusetts, 2003.

sured and estimated values. The average diameter of catbrier stems and branches was 0.23 cm (0.09 inches). Assuming vines are a perfect cylinder, this represents a surface-area-to-volume ratio of 17.5 cm^{-1} (535 ft^{-1}) for live and dead stems. Litter and live leaf material could not be directly measured, so we used a BEHAVEPLUS3-estimated value of 82.0 cm^{-1} ($2,500 \text{ ft}^{-1}$).

On average, 72% (range = 66–89%) of the fuel load was dead stems and litter, which tended to be concentrated in the lower 0.4 m (1.3 ft) of the fuel bed but were present throughout. The presence of dead stems in the upper parts of the fuel bed was due both to dead stems branching off of live stems and to dead, broken stems entwined in live catbrier. Dead catbrier leaves tend to curl as they dry, and this effect, coupled with small stems present in the litter layer, keeps the litter layer well aerated. The average depth of litter was 5.2 cm (2.0 inches).

The BEHAVEW module TSTMDL calculates the packing ratio based on component fuel loads and fuel bed depths. It further calculates a relative packing ratio by dividing the theoretical optimum packing ratio by the calculated observed packing ratio. The BEHAVE analysis for catbrier yielded a relative packing ratio of 0.79, which is close to the optimal value of 1.0.

Predicting Catbrier Fuel Load

Stem density, 3-dimensional cover sampling, and shrub height were sampled and evaluated independently as predictors of catbrier fuel load. Stem density was a poor predictor of fuel load ($R^2 = 0.06$), possibly due to the small sample size. The total number of intercepts (3-dimensional sampling) per quadrat predicted the total weight of fuels well ($R^2 = 0.80$) (Figure 1). There were, on average, 49 intercepts/quadrat, of which 27 (55%) were live and 22 (45%) were dead material. Live intercepts served as a good predictor of live weight, while dead intercepts were somewhat less useful as a predictor of dead weight ($R^2 = 0.73$ and

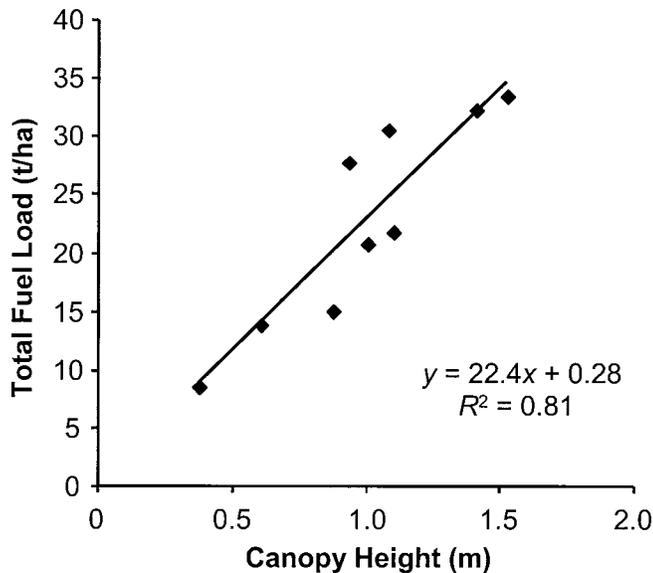


Fig. 2. Shrub canopy height as a predictor of total fuel load in catbrier fuel beds on Naushon Island, Massachusetts, 2003.

0.40, respectively). Average shrub height was the best predictor of the total weight of fuels, with $R^2 = 0.81$ (Figure 2). The average shrub height of catbrier was 1.00 m (3.27 feet), with a range of 0.38–1.53 m (1.25–5.02 ft). This parameter, which describes the overall fuel bed height, was also the most time-efficient method of the three for estimating the total weight of fuel.

Custom Fuel Model Development

We created a custom fuel model in the Windows-based BEHAVEPLUS3 and the DOS-based BEHAVEW using information from the fuel sampling portion of the project. We then used data from the prescribed fire to evaluate the model output and modify the input parameters. The two modeling programs differ in the way that data are entered and in the detail provided by their output, with BEHAVEPLUS3 having a simplified input interface requiring less information and an output report that omits predictions of packing ratio and packing ratio:optimum packing ratio. The two programs do, however, always yield identical predictions of flame length and rate of spread.

The components of the custom fuel model are presented in Table 2. No fuels larger than 0.64 cm (0.25 inches) basal diameter were present, so the loads for 10- and 100-h fuels were zero. In constructing this model, we considered all fuel particles in the leaf litter, including dead catbrier stems, to be litter. All fuels above the litter layer, including dead stems, live stems,

Table 3. Average observed (with range in parentheses) and predicted flame lengths and rates of spread for a catbrier custom fuel model (CFM) versus standard fuel models (SFM 3, 4, and 7), Naushon Island, Massachusetts, 2004.

Fuel model	Flame length (m)	Rate of spread (m/min)
Observed	5 (4.5–6)	11 (9–14)
Catbrier CFM	5	11
SFM 3 Tall Grass	3	24
SFM 4 Chaparral	2	6
SFM 7 Southern Rough	1	5

and live leaves were entered into the fuel model as part of the shrub component. The live shrub component is made up of all the live fuel particles in the fuel bed, including live leaves. A slash component was not used in this model. We used the standard heat content of 18,622 kJ/kg, which is widely used in fire modeling (Pyne et al. 1996) and is used for all 13 standard fuel models.

During the June 2004 prescribed fire, wind speeds averaged 16.1 km/h (range = 13–24 km/h), litter (non-woody) moisture content was 19%, 1-h dead wood was 8%, and live (leaf) fuels were 175%. Flame lengths averaged 4.6–6.1 m with rates of spread of 9.1–13.7 m/min. When time-of-the-burn fuel moisture and weather conditions are used with BEHAVEPLUS3, the custom fuel model predicts rates of spread and flame lengths that closely match observed fire behavior. The relevant standard fuel models all underpredict flame length and, with the exception of fuel model 3, underpredict rate of spread (Table 3).

Sensitivity Analysis

Rate of spread and flame length reacted differently to varying input parameters for the custom fuel model, with rate of spread generally changing more; i.e., a change in an input parameter usually caused the model to predict a greater percent change in rate of spread than in flame length. We calculated a combined sensitivity by averaging the sensitivity values for rate of spread and flame length (Table 4).

Generally, the custom fuel model was least sensitive to moisture of extinction and live fuel load. Decreasing the moisture of extinction by 40% resulted in a predicted change in flame length of 12%, whereas increasing the moisture of extinction by 40% resulted in no change in predicted flame length. Similarly, reducing the live fuel load by 40% resulted in no predicted change in flame length in the model, whereas increasing the live fuel load by 40% decreased flame length by only 6%.

Table 2. Custom fuel model inputs for a catbrier-dominated grassland, Naushon Island, Massachusetts, 2003.

Fuel component	Category	Size class	Load (t/ha)	SA:V ^a (cm ⁻¹)	Heat content (kJ/kg)	Moisture of extinction (%)	Fuel bed depth (m)
Litter	Dead	1 h	8.55	82	18,622	32	0.05
Shrub	Dead	1 h	7.96	17	18,622	32	1.00
Shrub	Live	1 h	6.28	32	18,622	32	1.00

^a Surface-area-to-volume ratio.

Table 4. Sensitivity of catbrier custom fuel model predictions to changes in tested fuel inputs. Sensitivity is measured as the ratio of the degree of change in the output (the predicted fire behavior parameter) to the degree of change in the input (the fuel parameter). Combined sensitivity is the average of the rate of spread and flame length sensitivities.

Fuel input	Combined sensitivity	Rate of spread sensitivity	Flame length sensitivity
Fuel bed depth	0.79	1.21	0.38
Surface area to volume	0.68	1.23	0.13
Heat content	0.55	0.47	0.62
1-h fuel load	0.30	0.29	0.31
Live fuel load	0.17	0.26	0.09
Moisture of extinction	0.13	0.15	0.11

The model was most sensitive to changes in surface-area-to-volume ratio and fuel bed depth. With all other fuel parameters held constant, increasing the fuel bed depth decreased the packing ratio and vice versa. The net result was a 45% increase in rate of spread with a 40% increase in fuel bed depth and a 45% decrease with a 40% decrease in fuel bed depth. Similarly, a 40% increase in the surface-area-to-volume ratio of dead fuels caused a 53% increase in predicted rate of spread, and a 40% decrease in dead fuel surface-area-to-volume ratio decreased predicted rate of spread by 42%.

DISCUSSION

Characteristics That Contribute to Extreme Fire Behavior

Several characteristics of catbrier fuels contribute to extreme fire behavior. Among these is the absence of a large fuel component that would act as heat sink and slow rates of spread. With no fuels >0.64 cm in diameter, more of the fire's energy is being expended in the flaming stage of combustion and little heat is being expended raising large fuel particles to the point of ignition. Nearly three-quarters (72%) of the fuel in catbrier fuel beds is dead, so little energy is used to drive water from live fuels.

Average 1-h dead fuel load of 16.5 t/ha (the sum of litter and dead stem fuels) is unusually large. None of the 13 standard fuel models has a fine dead fuel load larger than 11.4 t/ha. Although the surface-area-to-volume ratio of catbrier fuels is less than that of grass fuels, they are higher than those for most associated shrubs, including Scotch broom and huckleberry, which support lower rates of spread and flame lengths (Richburg et al. 2004).

High fuel loads distributed through a deep fuel bed contribute to the extreme fire behavior observed in catbrier. The proportion of the fuel bed that is occupied by fuel is defined as the packing ratio (Burgan and Rothermel 1984). A fuel bed with no fuel has a packing ratio of 0, and a solid block of wood has a packing ratio of 1 (Burgan and Rothermel 1984, Miller 1994). A very tightly packed (high packing ratio) fuel bed often will not burn well because of a lack of available oxygen. By contrast, a very loosely packed fuel bed

will similarly not burn well because fuel particles are spread so far apart that heat is not transferred readily among particles even though oxygen is readily available (Miller 1994). Every fuel bed has a theoretically ideal mix of fuel and air, and this mix is referred to as the optimum packing ratio (Burgan and Rothermel 1984). In catbrier fuel beds, fuel load and depth combine to produce a relative packing ratio of 0.79—a near ideal mix of fuel and air.

The way fuel is distributed within a fuel bed influences the rate of spread of a fire. Fires spread fastest in vertically and horizontally continuous fuels. Horizontal continuity is related to the horizontal distance between fuel particles, which is a function of percent cover, whereas vertical continuity is related to the distance between surface and crown fuels (Miller 1994). Our data show that both horizontal and vertical continuity of catbrier fuels is high.

Predicting Catbrier Fuel Load

A simple, efficient, and reliable method of predicting catbrier fuel load is necessary for reliable predictions of fire behavior and for documenting the effects of management on fuel beds. We evaluated three methods for estimating catbrier fuel loads and found canopy height to be the simplest and most reliable. Stem density predicts fuel load poorly, and counting stems is not practical without harvesting stems individually. Three-dimensional cover accurately predicts total fuel load, but sampling requires approximately 40 min/1-m² quadrat to perform. Attempts to correlate live intercepts with live fuel load and dead intercepts with dead fuel load were less successful because determining whether a catbrier stem was alive or dead without harvesting is difficult. Stand height, on the other hand, is both simple and efficient, and can be done with precision and accuracy. Sampling can be performed quickly with minimal equipment (a stick for measuring height and brush chaps for walking through the thorny vines).

Total fuel load has little value in predicting fire behavior; predictions are more accurate when fuel load is broken down by category (shrub, slash, litter, grass), size class (1-, 10-, or 100-h), and status (live or dead). The catbrier fuel beds we sampled comprised exclusively shrub and litter components and were all <0.64 cm in diameter (i.e., 1-h fuels). A variation of our 3-dimensional cover sampling, where fewer points are sampled, might facilitate the determination of the proportion of live and dead shrub stems.

Utility of Custom Fuel Models

Our custom fuel model accurately predicted fire behavior observed during a June 2004 prescribed burn. Custom fuel model predictions were superior to those produced by the standard fuel models (Table 3), which, for the most part, underpredicted observed flame length and rate of spread. In a related study, Richburg et al. (2004) compared observed versus custom fuel model-predicted flame lengths for a variety of shrubs and found a correlation coefficient (*r*) of 0.96. The fact

that these custom fuel models predict fire behavior well, coupled with the fact that standard fuel model predictions were generally much poorer, supports the use of custom fuel models for catbrier. The June 2004 prescribed fire was useful in confirming the accuracy of the catbrier custom fuel model, and the extreme fire behavior was effective in showing the potential danger of this fuel condition.

Identifying Important Fuel Bed Characteristics

By evaluating how variations in input values affected the flame length and rate of spread predicted by our custom fuel model, we were able to determine which fuel bed characteristics were most important in affecting fire behavior. This knowledge is useful to managers wishing to modify fuel beds to reduce fire hazard. Both parameters were most sensitive to changes in fuel bed depth and least sensitive to moisture of extinction. The former is easily manipulated in the field (as by chopping or flail-mowing—see Richburg et al. [2004]), whereas moisture of extinction depends not only on fuel arrangement but on the thermodynamic properties of individual fuel components.

Surface-area-to-volume ratio and heat content both strongly influence rate of spread and flame length, but both are difficult to measure. Although we directly measured catbrier stem surface-area-to-volume ratio, we had to approximate the ratio for the litter component. Additional measurements of catbrier heat content are needed. The live leaves of catbrier appear waxy, suggesting the presence of volatile compounds that should yield a higher heat content than those without waxy leaves (Burgan and Rothermel 1984). Neither surface-area-to-volume nor heat content are easily manipulated, however, and therefore should not be a focus of management efforts.

Although 1-h size-class fuel load has less effect on rate of spread than most fuel characteristics, it is next in importance (after heat content and fuel bed depth) in its effect on flame length. Coefficients of determination suggest that we can accurately predict 1-h fuel load with both 3-dimensional cover sampling and by measuring shrub heights. High live fuel loads, because of their high moisture content, tend to suppress fire behavior. This effect is demonstrated by the sensitivity analysis. Increasing the live fuel load in BEHAVE has the effect of reducing both rate of spread and flame length, and decreasing live fuel load increases fire behavior. But catbrier fuels are unique among northeastern U.S. fuel complexes in their ability to support extreme fire behavior with live fuel moistures in excess of 150%, as our June 2004 prescribed burn illustrated.

MANAGEMENT IMPLICATIONS/AREAS FOR FURTHER STUDY

Fire managers working in the Atlantic Coastal Plain have long recognized the extreme fire hazard associated with catbrier-dominated landscapes. We have, for the first time, quantitatively described the fuel bed

characteristics associated with this hazard and identified components that might be most effectively manipulated to reduce problem fire behavior. Our fuel sampling protocol and the custom fuel model we developed will allow managers to more effectively manage catbrier fuels and evaluate outcomes in the context of altered fire behavior. Additional sampling in other areas dominated by catbrier is necessary to determine if the catbrier fuel bed is similar among sites. Although difficult to implement, additional research prescribed burns in catbrier monocultures will increase our knowledge of how fire behavior is affected by variations in environmental conditions. Our one research burn was conducted under moderate fire weather conditions during the growing season and resulted in impressive fire behavior. To more fully test our custom fuel model, we need additional documentation of burns under other conditions, especially those in the dormant season. Only with additional research will we be able to provide fire managers with the tools that they need to effectively manage this hazardous fuel type.

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FIRE BEHAVIOR ASPECTS ASSOCIATED WITH LINEAR DISTURBANCES IN ALBERTA, CANADA

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ABSTRACT

Recently, a number of large spring fires have highlighted the fire-suppression problems created by linear disturbances in Alberta, Canada. Grass commonly colonizes these areas and, when in a cured condition, this fine fuel readily contributes to the ease of ignition, rapid rates of fire spread, and overnight fire growth. The Forest Engineering Research Institute of Canada's (FERIC) Wildfire Operations Research Group is investigating this issue by documenting seasonal variations in fuel loads and developing a simple method for readily estimating fuel loads, testing the effectiveness of fuel breaks composed of selected fire-resistant vegetation, examining the effects of mowing as a fuel modification technique, and assessing the effectiveness of firebreak widths in relation to burning conditions.

keywords: Alberta, Canadian Forest Fire Danger Rating System, firebreak, fire intensity, fire-resistant vegetation, fuel break, fuel load, fuel modification.

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INTRODUCTION

The increasing activity in the oil and gas industry in Alberta, Canada, has led to extensive development in the forests of central and northern Alberta. The most visual impact of these developments has been the creation of linear disturbances: powerline rights-of-way, pipelines, seismic lines, roads, and railroads, which range in width from 6 to 200 m. These man-made corridors cutting through the forested areas have made the task of fire suppression more difficult on a number of large wildfires that have occurred in Alberta during the last 10 y. When these disturbances are created, grass (commonly marsh reed or bluejoint [*Calamagrostis canadensis*] in central and northern Alberta) quickly invades the site, and this abundant fine fuel, when in a fully cured state, dries out very quickly after rain, greatly increasing the potential for explosive fire spread. The scope of the problem is sizeable in Alberta and continues to grow with the amount of resource exploration taking place and the resulting linear disturbances crisscrossing Alberta's forests.

When grass is in a cured condition such as in the spring (March–early June) and fall (September–October), or even in winter if no snow is present, it exhibits the following attributes with respect to wildland fire potential: 1) ignition is relatively easy, 2) rapid rates of fire spread are possible, and 3) the fuel beds are capable of carrying fire long distances overnight by “wicking” at a time when firefighting resources are typically reduced.

The fire hazard associated with linear disturbances is a priority for many of Forest Engineering Research Institute of Canada's (FERIC) members who have re-

quested an investigation into techniques to mitigate the potential for large fire occurrence associated with these strips of land. Many values-at-risk within Alberta's forests are located on or in close proximity to linear disturbances, and these would obviously benefit from a reduction in current fire hazard levels.

FERIC's Wildland Fire Operations Research Group is investigating both fire behavior and techniques to reduce fire behavior potential along linear disturbances. This includes a fuel load survey, assessing the feasibility of establishing strips of fire-resistant vegetation, and examining the effectiveness of mowing in reducing potential fire behavior. Firebreak breaching in relation to the fire environment is also being studied.

FUEL LOAD ASSESSMENT

An accurate knowledge of existing fuel loads associated with linear disturbances could help managers make more reliable predictions of fire behavior, for both strategic planning and for near-real operational purposes. Currently a default or nominal value is used for fuel load in the grass fuel types of the Canadian Forest Fire Behavior Prediction (FBP) System (Forestry Canada Fire Danger Group 1992) to calculate fire intensity, which is in turn related to flame length (Byram 1959):

$$I = Hwr/600,$$

where I = fire intensity (kW/m), H = low heat of combustion (kJ/kg), w = available fuel consumed (t/ha), and r = rate of fire spread (m/min). The relevancy of this assumed value for fire behavior predictions in linear disturbance fuel types is presently un-

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Fig. 1. Using the grass disk meter to estimate grass fuel load near Whitecourt, Alberta, May 2005.

known. Therefore, FERIC is quantifying the fuel loads in a study with three primary objectives: 1) create a database of fuel loads sampled along various linear disturbances in central and northern Alberta, 2) determine a mean fuel load for use in predicting fire behavior in grass fuel beds on linear disturbances in both the spring and the fall fire seasons, and 3) develop a simple field technique for readily estimating grass fuel loads.

During the summer of 2005, FERIC collected fuel load samples along linear disturbances in central and northern Alberta in conjunction with a “grass disk meter” sampling technique originally developed and calibrated in South Africa (Trollope and Potgieter 1986). This technique involves correlating the settling height of the round disk on the grass fuel bed (Figure 1), with the sampled fuel load compressed by the disk’s weight (Figure 2). The samples were separated into 5-cm height classes, and mean weights were determined for each class per Trollope and Potgieter (1986).

A total of 64 grass fuel load samples were collected from pipelines, powerline rights-of-way, and seismic lines in May 2005. The locations were recorded and photographed, and resampled in September 2005. The separate fuel load values determined for spring and fall conditions will, in turn, be used to calculate fire intensities for use during going wildfire events or in simulations.

The mean fuel loads during the spring and fall collections were 3.5 t/ha (SD = 1.7) and 5.4 t/ha (SD = 3.5), respectively. In comparison, the Canadian FBP System grass fuel type models use a default value of 3.0 t/ha. This 0.5-t/ha fuel load difference in the spring corresponds to a slight increase (12%) in fire intensity but quite a significant increase (80%) in the fall because the fuel load was nearly double.

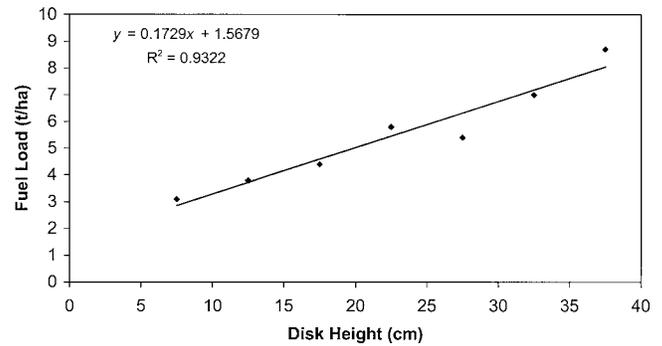


Fig. 2. Relationship between disk height and fuel load for both spring (March–early June) and fall (September–October) fuel load samples collected in Alberta in 2005.

FUEL/VEGETATION TYPE CONVERSION

One fuel management technique practiced along linear disturbances involves the planting of less flammable vegetative cover to reduce potential fire ignition and spread. The use of such fire-resistant plants has been studied in Australia, New Zealand, the United States, and in Ontario, Canada, for Bell Canada and CP Rail (Hogenbirk 1996). Several large fires in Alberta in recent years have sparked interest in planting or seeding less flammable vegetation in strategic locations along linear disturbances to minimize the potential for fire spread and, in turn, large fire growth. The goal of this particular FERIC study is to determine if strips of selected vegetative species established along linear disturbances can significantly reduce fire ignition and spread potential.

Less flammable vegetation may have certain desirable characteristics, but the introduction of nonnative species to different locations of the province is a regulated process. The Alberta Department of Sustainable Resource Development has enforceable guidelines on the use of “introduced” species. Thus, any species chosen for testing must be native to the province.

Many fire-resistant plant species tend to have similar physical fuel characteristics. Hogenbirk (1996) found the following traits related to fire resistance, in order of importance: 1) stem density, 2) radius of stems, 3) surface-area-to-volume ratio, and 4) moisture content. These traits have been measured and their relationships to plant ignitability have been determined (Hogenbirk 1996). Surprisingly, moisture content was rated the fourth most important variable. Hogenbirk (1996) also compiled a list of characteristics of the ideal species for “greenstripping,” which is the use of less flammable plants to reduce fire hazard. These plants should 1) reduce the probability of ignition throughout the year, 2) reduce fire spread rates, 3) remain dominant on a site for at least 10 y, and 4) create no additional environmental or safety hazards.

The plants should also 1) grow quickly in the spring, 2) maintain a high moisture content, 3) be low growing, 4) produce small quantities of dead standing crop, 5) produce small amounts of litter, 6) decompose quickly as litter, 7) out-compete other plants, 8) rees-



Fig. 3. A plot established for ignition testing on the grounds of the Vegreville branch of the Alberta Research Council, June 2006. The plot shown contains white clover and is 4×4 m in area.

establish dominance following disturbance, and 9) be cost-effective. It was considered to be most beneficial to find species that green quickly in the spring and are low growing. If green-up could occur even 10 d earlier than normal, the length of the spring fire season would correspondingly be greatly decreased.

A literature search and interviews with vegetation management specialists was accordingly undertaken to identify plants native to Alberta meeting the above criteria. Six species were selected for testing: white clover (*Trifolium repens*), alsike clover (*Trifolium hybridum*), yarrow (*Achillea millefolium*), fireweed (*Epilobium angustifolium*), crested wheatgrass (*Agropyron cristatum*), and sheep fescue (*Festuca ovina*).

FERIC is cooperating with a unit of the Alberta Research Council (ARC) located in Vegreville, Alberta, in this study. Test plots were established and planted on ARC experimental grounds in the spring of 2005 (Figure 3). Plans call for ignition trials and fire behavior studies in the spring and fall of 2006 (if sufficient vegetative growth occurs over the intervening period). Six small plots (4.0×4.0 m) will be used for ignition trials, with 1.0×1.0 -m subplots ignited during the spring, summer, and fall. One large plot (100×100 m) will be used for testing resistance of the vegetative cover to high-intensity flame front. In the latter case, a "line of fire" will be ignited in standing, cured grass and allowed to run up to a 40-m-wide "barrier" consisting of the six species selected for testing. This effectiveness of this fuel break will be observed and documented (e.g., penetration depth). Based on these field tests, the most successful species may then be examined for their operational suitability on existing linear disturbances.

FUEL MODIFICATION

The mowing of grass along linear disturbances is a common fuel modification technique used to reduce both the grass fire hazard and the development of large



Fig. 4. Difference in fire behavior between fire in unmowed or natural grass (background) and mowed grass (foreground), Fort McMurray, Alberta, May 2005. Both areas were ignited at approximately the same time.

er fuels (e.g., woody shrubs and tree reproduction), but little research has been done to validate its effectiveness. The objective of this aspect of the FERIC project on linear disturbances is to determine how mowing influences fire behavior in treated areas in comparison to untreated or modified areas. To document this, experimental fires were carried out during the 2004 and 2005 fire seasons near the communities of Fort McMurray in northeastern Alberta and Slave Lake in central Alberta.

The experimental fires involved simultaneous "line fire" ignitions in treated (i.e., mowed) and untreated (i.e., natural grass) plots (Figure 4). Each plot measured 50×50 m in area. Fire behavior characteristics (i.e., forward spread and flame length) were observed and recorded during the fires. Six pairs of experimental fires have been completed to date, involving four spring comparisons and two fall comparisons. Six more plot comparisons are planned.

In the spring of 2004, the average rate of spread (ROS) in recently mowed plots was approximately 15% less than that attained in the natural or unmowed grass; this also held true during wind gusts. For spring burning in 2005, the plots near Fort McMurray were mowed during the previous fall and allowed to settle over the winter. This resulted in lighter fuel loads, a quicker green-up and, in turn, a substantial reduction in fire spread compared to the previous spring's results, as the average ROS in the mowed plots was only about 20% of that obtained in the unmowed or natural grass plots. All clippings were left in place following mowing.

The experimental fires carried out to date may very well illustrate the differences in fire behavior resulting from the timing of the mowing treatments. Although only two trials involving fall comparisons have been completed, a significant difference was observed in fire behavior in the mowed and natural grass plots compared to the fire behavior observed in the mowed plots during the spring of 2004 at the same sites. This raises the issue of the timing of the treatment and/or repeated burns. More comparison trials are planned in

2006 involving both fall and spring mowing in order to better understand the implications of treatment timing on fire behavior. It is thought that fall precipitation acts to speed the decomposition process, and thus lighter fuel load, and that the winter snowpack compresses the remaining fuel, thereby reducing the fire spread and intensity potential. Fuel loads in the plots that had been mowed in the fall were half those of spring mowing, which would result in a 50% reduction in fire intensity.

Observed flame lengths on the natural grass plots were almost 3 times greater than in the mowed plots (Figure 4). This is a critical bit of knowledge, as flame length is one of the major determinants of the likelihood of successful containment by suppression forces (Byram 1959).

FIREBREAK BREACHING

Because spotting in grass fires in Alberta is not common, firebreaks are typically able to reduce or halt fire spread. In addition to investigating the use of less flammable plant species as fuel breaks, FERIC is also examining the optimum width for firebreaks in Alberta's fire environments.

As part of this study, FERIC is attempting to determine the minimum firebreak width required to control the development of grass fires in Alberta during its spring and fall fire seasons. Toward this end, FERIC is analyzing historic fire weather databases and has developed a software package for gauging the effectiveness of firebreaks in stopping grassland fires based on coupling existing models for firebreak breaching developed by Wilson (1988) with the two major modules or subsystems of the Canadian Forest Fire Danger Rating System (CFFDRS)—i.e., the Fire Weather Index System (Van Wagner 1987) and the FBP System. The rate of fire spread output from the FBP System is being combined with the results from the grass fuel load survey carried in central and northern Alberta in order to estimate fire intensities. Wilson's (1988) models output the probability of firebreak breaching based on fire intensity and firebreak width.

Once the optimum width of an effective firebreak or fuel break has been approximated, the next step will be to apply this knowledge at the landscape level. As it would be impossible both physically and economically to treat all linear disturbances in the province,

any treatments applied must be strategically placed in order to be both cost-effective and efficient.

PROMETHEUS, an existing landscape-level fire behavior and growth model based on the CFFDRS (Tymstra 2002), permits the user to alter fuel types and insert firebreaks or barriers to fire spread of varying widths, and then to undertake simulated fire runs over the landscape. This will allow determination of the best alignment of firebreaks and fuel breaks in order to minimize problematic fire behavior associated with linear disturbances.

CONCLUSION

A wide variety of industrial and other forest users in northern and central Alberta stand to benefit from strategies that reduce problematic fire behavior associated with linear disturbances and, in turn, contribute to containing wildfires to the smallest size possible. FERIC is currently undertaking research into four facets of fire behavior associated with linear disturbances, with a view to reducing the potential for adversely impacting the task of controlling wildfires. As more definitive results become available, the findings will be published in FERIC Advantage Reports and made available for downloading on the Internet (<http://fire.feric.ca/36202003/36202003.asp>).

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QUANTIFYING FIRE BEHAVIOR VERSUS SOCIETAL BENEFITS OF SOUTHERN CALIFORNIA SHRUBLANDS AND GRASSLANDS

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ABSTRACT

Urban sprawl in southern California perpetually threatens native shrublands and grasslands, which intrinsically provide both biophysical and socioeconomic benefits to society. However, these vegetation types are simultaneously prone to high-intensity wildfires that lead to enormous damage to human interests. After the southern California firestorms of October 2003, new regulations were adopted that increased the mandatory vegetation clearance around structures in order to reduce fire risk, which may significantly impact the positive benefits that grasslands and shrublands provide. To address this apparent conflict, we investigated the tradeoffs between societal benefits derived from major shrubland, grassland, and woodland vegetation types in southern California versus the potential fire behavior associated with each vegetation type.

Two state-of-the-art, geographic information system-based software packages were utilized in the analysis, which focused on San Diego County, California. For each of the most common grassland, shrubland, and woodland vegetation communities in the area, FARSITE was utilized to assess potential fire behavior under average and extreme weather conditions. The most extreme fire behavior was found in nonnative grasslands and scrub oak chaparral communities and least extreme in coast live oak (*Quercus agrifolia*) communities. Under Santa Ana wind conditions, simulated fires in almost all vegetation types burned over 3 km into a developed area in <1 h. CITYgreen was used to quantify air pollution removal, carbon sequestration, and stormwater retention for each of the vegetation types, but was found to be largely ineffective because it calculated no measurable benefits for any non-tree vegetation types. To ensure sustainable neighborhoods in the wildland-urban interface, diverse stakeholders must create collaborative management plans that simultaneously reduce fire risk and maximize societal benefits.

keywords: chaparral, CITYgreen, FARSITE, fire behavior modeling, grassland, shrubland, southern California, wildland-urban interface.

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INTRODUCTION

In the wildland-urban interface, differing vegetation types provide various levels of tangible and intangible benefits to society. For example, vegetation not only enhances community attractiveness, but also lowers home cooling costs (Taha et al. 1997), reduces air pollution (Taha 1996, Taha et al. 1997) and stormwater runoff (Sanders 1986), and sequesters carbon (Nowak and Rowntree 1991, McPherson et al. 1994). However, vegetation may simultaneously increase the risk to human development because it fuels wildfires. The type and structure of a given vegetation community will inherently influence both the benefits and the risk to a local development.

This seeming paradox in societal benefits versus fire risk of vegetation is readily exemplified in southern California, where a burgeoning population has regularly developed adjacent to and intermixed with highly fire-prone shrubland and grassland communities. The population of San Diego, Los Angeles, Orange,

Riverside, San Bernardino, and Ventura counties in southern California in 2000 was 20.5 million and is expected to grow by at least 10% over the next 10 y (data from U.S. Census Bureau), which will continue to cause an enormous conversion of native vegetation to developed areas. For example, from 1985 to 2002, the city of San Diego experienced a 39% increase in urban areas across the landscape, which led to a 32% loss of grasslands, 7% loss of shrublands, and 27% of loss of tree cover (American Forests 2003).

While the region's shrubland, grassland, and woodland communities provide numerous benefits, they are also prone to high-intensity, destructive wildfires. For example, the 2003 Cedar Fire in San Diego County, the largest and most destructive fire in California's history, burned across 273,246 acres, killed 24 people, and destroyed 4,847 structures (California Department of Forestry & Fire Protection 2004). As a result of the 2003 Fire Siege, California Senate Bill 1369 was signed into law in 2004, which amended Public Resources Code 4291 to increase mandatory vegetation clearance around homes from a previous standard of 9.14 m (30 ft) to a current standard of

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Table 1. Holland (1986) and CITYgreen software classifications of major vegetation communities in San Diego County, California.

Holland vegetation classification	Holland description	CITYgreen classification
42110–Foothill Grassland	Perennial purple tussockgrass (<i>Nassella pulchra</i>) or needlegrass (<i>Stipa</i> spp.) to 0.6 m with interspersed annuals	Pasture/Range (continuous forage for grazing): ground cover >75%
42200–Non-native Grassland	Dense annual grasses with flowering culms to 1 m	Pasture/Range (continuous forage for grazing): ground cover >75%
37900–Scrub Oak Chaparral	Dense scrub oak (<i>Quercus berberidifolia</i>), coastal sage scrub oak (<i>Q. dumosa</i>), birchleaf mountain mahogany (<i>Cercocarpus montanus</i> var. <i>glaber</i>) to 6 m	Shrub: ground cover >75%
71160–Coast Live Oak Woodland	Coast live oak (<i>Quercus agrifolia</i>) 10–25 m with understory toyon (<i>Heteromeles arbutifolia</i>), currants (<i>Ribes</i> spp.), laurel sumac (<i>Malosma laurina</i>), or blue elderberry (<i>Sambucus nigra</i> ssp. <i>canadensis</i>)	Trees: forest litter understory: no grazing, forest litter and brush adequately cover soil
37200–Chamise Chaparral	Chamise (<i>Adenostoma fasciculatum</i>) to 3 m with little herbaceous understory	Arid & Semi-Arid Rangeland: desert shrub: ground cover between 40% and 70%
37120–Southern Mixed Chaparral	Coastal sage scrub oak, chamise, manzanita (<i>Arctostaphylos</i> spp.), ceanothus (<i>Ceanothus</i> spp.) 1.5–3 m with little understory	Arid & Semi-Arid Rangeland: desert shrub: ground cover >70%
32501–Diegan Coastal Sage Scrub	California sagebrush (<i>Artemisia californica</i>), California buckwheat (<i>Eriogonum fasciculatum</i>), white sage (<i>Salvia apiana</i>)	Arid & Semi-Arid Rangeland: sagebrush: ground cover between 40% and 70%

30.48 m (100 ft) in all designated areas where the state has the primary suppression responsibilities. These new standards have the potential to significantly reduce the losses caused by wildfire, but will also likely reduce the many tangible benefits to society that the vegetation provides.

To assist decision-making by land managers in the wildland–urban interface of southern California, we utilized two widely used geographic information system (GIS)–based applications to explore the tradeoffs in quantifiable benefits to society versus the inherent fire risk of major shrubland, grassland, and woodland community types in San Diego County. Our major objectives were to 1) quantify wildfire rate of spread, flame length, and fireline intensity under both average and extreme weather conditions for 7 major shrubland, grassland, and woodland community types that are common in San Diego County; 2) quantify stormwater runoff removal, air pollution reduction, and carbon sequestration for each of those same vegetation types; and 3) evaluate each of the major vegetation types for minimization of fire behavior and maximization of societal benefits.

METHODS

Our analysis employed FARSITE 4.1.03 (USDA Forest Service Fire Sciences Laboratory, Missoula, MT) for fire behavior simulations and CITYgreen for ArcGIS (American Forests, Washington, D.C.) for calculation of societal benefits. GIS layers necessary for the analysis were obtained from the San Diego Association of Governments and included a 10-m digital elevation model (DEM) and a vegetation classification shape file. Those layers were converted into forms required by FARSITE and by CITYgreen with the Spatial Analyst extension in ArcMap 9.2. FARSITE required ASCII data for elevation, slope, aspect, fuel model, and canopy coverage. The Spatial Analyst ex-

tension of ArcMap derived the slope and aspect grids from the DEM and then used the DEM as a background basis for deriving the fuel model grid from the vegetation shape file. The ArcToolbox functions in ArcMap were then used to create ASCII files from these raster data sets. For CITYgreen analysis, the vegetation shape file was converted to a grid with Spatial Analyst.

Vegetation communities in San Diego County were classified per Holland (1986) and categorized by CITYgreen protocol to calculate societal benefits (Table 1). Each vegetation community had previously been assigned a standard or custom fire behavior fuel model for pre-fire planning purposes in the San Diego area (M. Scott, Rancho Santa Fe Fire District, personal communication); these fuel model designations were used in the present study. Using ArcMap 9.2, we determined the most prevalent shrubland, grassland, and woodland fuel models in San Diego County to be standard fuel models 1, 3, 4, and 9 (Anderson 1982), and southern California custom fuel models SCAL15, SCAL17, and SCAL18 (see BehavePlus 3.0.2 for specific values associated with southern California custom fuel models). Within each of those fuel models, the most prevalent vegetation community, by area, was selected to represent vegetation classification per CITYgreen protocol. For a given CITYgreen analysis, all vegetation across the landscape was converted to a single type.

A “typical” area in San Diego County was then sought in which to analyze potential fire behavior and societal benefits for each of the pertinent vegetation communities. The Scripps Ranch (32°54.134'N, 117°05.985'W), a housing community in the wildland–urban interface of inland San Diego County, was chosen for the analysis because it had the majority of pertinent vegetation communities in or near the vicinity and also had 322 homes that were consumed during the 2003 Cedar Fire. A subsection of land adjacent to

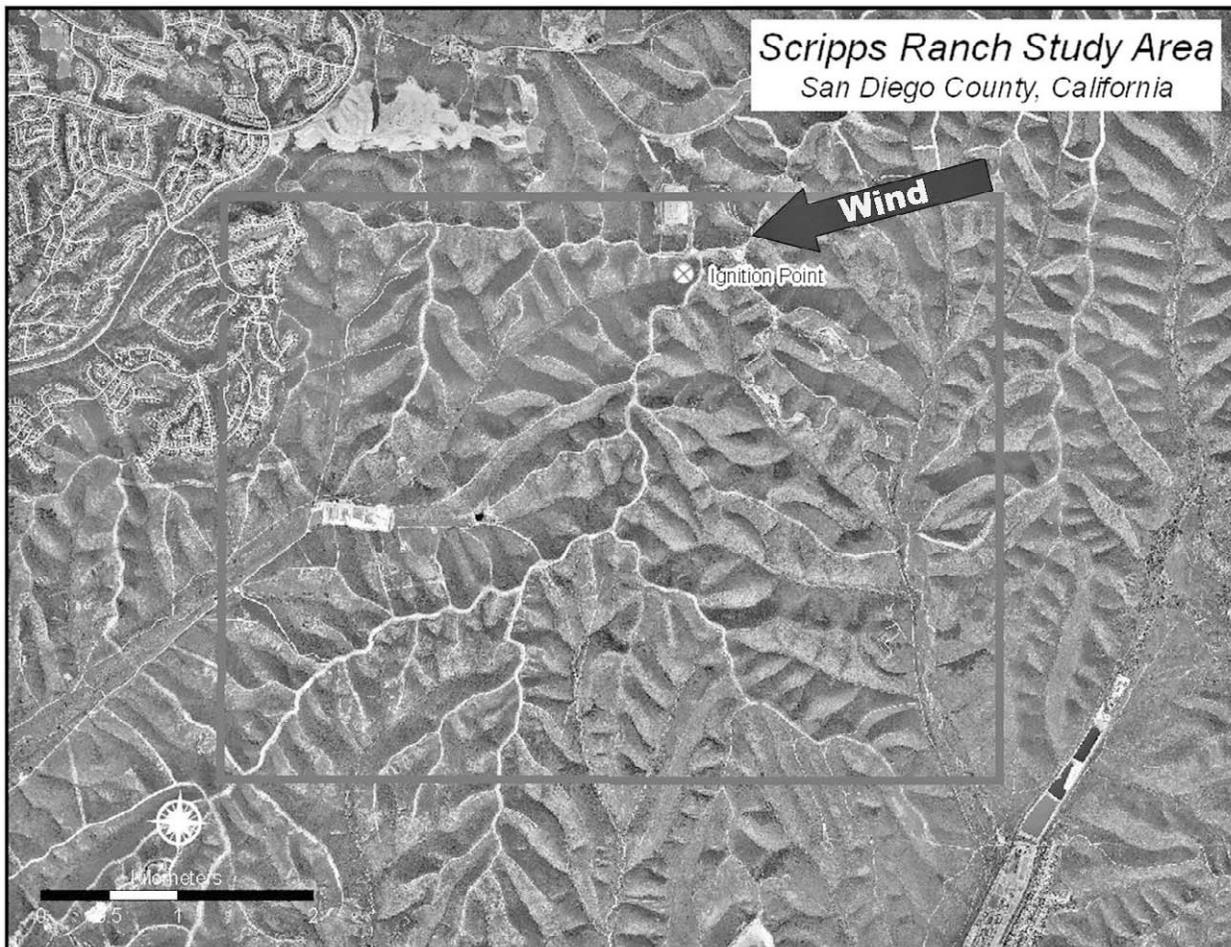


Fig. 1. Aerial photograph of the Scripps Ranch study area, San Diego County, California. Outline designates the area in which societal benefits of vegetation communities were calculated by CITYgreen software. The ignition point represents a potential location for a human-caused ignition. The arrow shows the characteristic direction of local Santa Ana winds. Photo taken in 2005 by Air-PhotoUSA.

and including a small portion of the Scripps Ranch (7,742 ha) was utilized in the analysis (Figure 1). For fire-simulation purposes, a single potential ignition point was designated at the junction of two major secondary roads, which, due to relative ease of access,

Table 2. Average and extreme weather and fuel moistures during September–October for FARSITE fire behavior simulation in San Diego County, California.

Weather parameter ^a	Percentile	
	Average 50%	Extreme 97%
High temperature (°C)	28.3	37.8
Relative humidity (%)	45	12
Wind speed (km/h)	12.9	64.4 ^b
Fuel moisture (%)		
1-h fuels	7	3
10-h fuels	9	4
100-h fuels	16	11
Live herbaceous fuels	93	5
Live woody fuels	1	1

^a Data from Poway RAWS, San Diego, California, September–October, 1981–1997.

^b Winds changed from calculated 21 km/h to reflect Santa Ana conditions.

was meant to reflect a likely location for a human-caused ignition.

Weather data required for FARSITE analysis was calculated by FireFamilyPlus 3.0.5 (Systems for Environmental Management and USDA Forest Service Fire Sciences Laboratory, Missoula, MT) with historic weather data (1981–1997) from the nearby Poway RAWS (Remote Automated Weather Station). Pertinent weather and fuel moisture data (high temperature, minimum relative humidity, wind speed, as well as 1-h, 10-h, 100-h, live herbaceous, and live woody fuel moistures) were calculated for average (50th percentile) and extreme (97th percentile) fire weather during the months of September and October, the two months that have historically burned most frequently in the area (Table 2). For each FARSITE simulation, temperature and relative humidity were kept constant throughout the day. Winds were also held constant throughout the day for both weather percentiles and were from north 67° east (characteristic direction of local Santa Ana winds); however, because calculated 97th percentile winds did not accurately reflect the most extreme fire conditions, namely Santa Ana winds, 97th percentile

Table 3. FARSITE mean fire behavior outputs under various weather scenarios for major vegetation communities (Holland 1986) in San Diego County, California.

Fuel model	Holland vegetation classification	Treatment		Fire behavior output				
		Weather (%)	Wind (%)	Rate of spread (m/min)	Flame length (m)	Fireline intensity (kW/m)	Area (ha)	Perimeter (km)
1	Foothill Grassland	50	50	5.9 (0.02) ^a	0.6 (0.020)	92.1 (0.29)	501.4	10.1
		97	50	9.1 (0.02)	0.8 (0.001)	182.3 (0.32)	508.4	10.2
		97	97	23.6 (0.06)	1.2 (0.001)	489.3 (1.29)	1,981.2	38.7
3	Non-native Grassland	50	50	13.8 (0.02)	2.4 (0.001)	1,850.7 (2.19)	2,268.0	41.6
		97	50	20.3 (0.02)	3.2 (0.002)	3,535.5 (3.12)	2,300.5	43.0
		97	97	25.2 (1.29)	3.6 (0.009)	5,138.2 (20.70)	1,673.8	31.9
4	Scrub Oak Chaparral	50	50	10.5 (0.02)	3.8 (0.003)	4,976.0 (8.07)	822.0	17.3
		97	50	13.5 (0.17)	4.6 (0.003)	7,638.8 (10.17)	960.3	19.6
		97	97	19.3 (0.09)	5.3 (0.013)	11,791.0 (50.4)	1,252.4	27.7
9	Coast Live Oak Woodland	50	50	0.7 (0.001)	0.4 (0.003)	45.9 (0.61)	7.7	1.0
		97	50	1.0 (0.010)	0.6 (0.003)	92.5 (0.86)	7.8	1.0
		97	97	6.8 (0.03)	1.4 (0.003)	588.7 (2.38)	189.1	6.0
15	Chamise Chaparral	50	50	2.3 (0.01)	1.4 (0.005)	605.3 (3.90)	46.8	2.5
		97	50	4.7 (0.02)	2.2 (0.004)	1,585.3 (5.14)	121.9	4.4
		97	97	8.7 (0.04)	2.7 (0.006)	2,958.6 (14.32)	388.4	15.1
16	Southern Mixed-Chaparral	50	50	2.9 (0.02)	1.6 (0.006)	787 (5.5)	100.7	3.9
		97	50	11.2 (0.02)	3.7 (0.003)	4,871.7 (7.50)	685.7	13.3
		97	97	13.3 (0.06)	4.1 (0.013)	6,391.7 (29.84)	801.8	20.3
18	Diegan Coastal Sage Scrub	50	50	4.3 (0.01)	2.8 (0.005)	2,564.0 (9.0)	160.3	5.1
		97	50	7.8 (0.02)	4.1 (0.004)	5,955.7 (11.7)	331.8	7.7
		97	97	9.8 (0.05)	4.3 (0.012)	7,882.9 (37.4)	528.0	16.4

^a Parenthetical values represent the standard error of means.

winds were changed from the calculated 21 km/h to a more reflective 65 km/h. Fuel moistures for both weather percentiles were input into a required fuel moisture file in a FARSITE project.

Fires were simulated for multiple combinations of fuel models and weather scenarios. For a given simulation, vegetation across the landscape was converted to the single fuel model of interest; nonburnable models remained nonburnable. Each landscape of a single fuel model was then simulated under three separate weather scenarios, including 1) 50th percentile weather, 50th percentile winds (average conditions); 2) 97th percentile weather, 50th percentile winds (extreme temperature, relative humidity, and fuel moisture without Santa Ana winds); and 3) 97th percentile weather, 97th percentile winds (extreme temperature, relative humidity, fuel moisture, as well as Santa Ana winds). Simulation parameters were as follows: Time Step = 30.0 min, Visible Time Step = 1.0 h, Perimeter Resolution = 30 m, Distance Resolution = 30 m. The conditioning period for fuel moistures was 1 d. Fires were simulated for 3 h, which allowed adequate demonstration of fire spread across the landscape while confining fire boundaries to the study area (during non-Santa Ana conditions). Output ASCII raster files of rate of spread, flame length, and fireline intensity were created for each simulation. After deleting all nonburned observations in the output ASCII raster layers, a general linear models procedure was conducted in the SAS System for Windows 8.02 (SAS Institute, Cary, NC) to test significance between fuel models for each of the three weather scenarios.

RESULTS

All fire behavior parameters varied significantly between vegetation types for each of the three weather

scenarios (all $P < 0.001$) (Table 3). The fastest rate of spread was in the nonnative grass community, followed by scrub oak chaparral, Diegan coastal sage scrub, and foothill grass communities. The highest flame lengths and fireline intensities were observed in the scrub oak chaparral, followed by the Diegan coastal sage scrub and the nonnative grass communities. Southern mixed-chaparral demonstrated the greatest range in variation in simulated fire behavior between weather scenarios.

Under normal weather conditions, only fires in the nonnative grasslands and scrub oak chaparral reached the homes within 3 h (Figure 2). As expected, fire behavior increased dramatically during extreme weather, particularly when the Santa Ana winds were simulated. Under Santa Ana conditions, the simulated fires arrived at the subdivision within 1 h in all vegetation types except for the coast live oak (*Quercus agrifolia*) type (Figure 3). It should be noted, however, that differences in flame length between vegetation types could potentially affect the survivability of the homes, based on construction techniques and materials.

Because the simulated fires extended beyond the project area during Santa Ana conditions in all vegetation communities except coast live oak (Figure 3), the calculated areas and perimeters in those communities were underestimated. Indeed, the fire simulated during Santa Ana conditions in nonnative grassland actually burned less area than during less extreme conditions (Table 3) because of a wind-induced increase in the fire's length-to-width ratio and a boundary that extended beyond the project area (Figure 3).

Only the coast live oak vegetation type showed any tangible societal benefits. In the coast live oak scenario, CITYgreen calculated that the trees removed

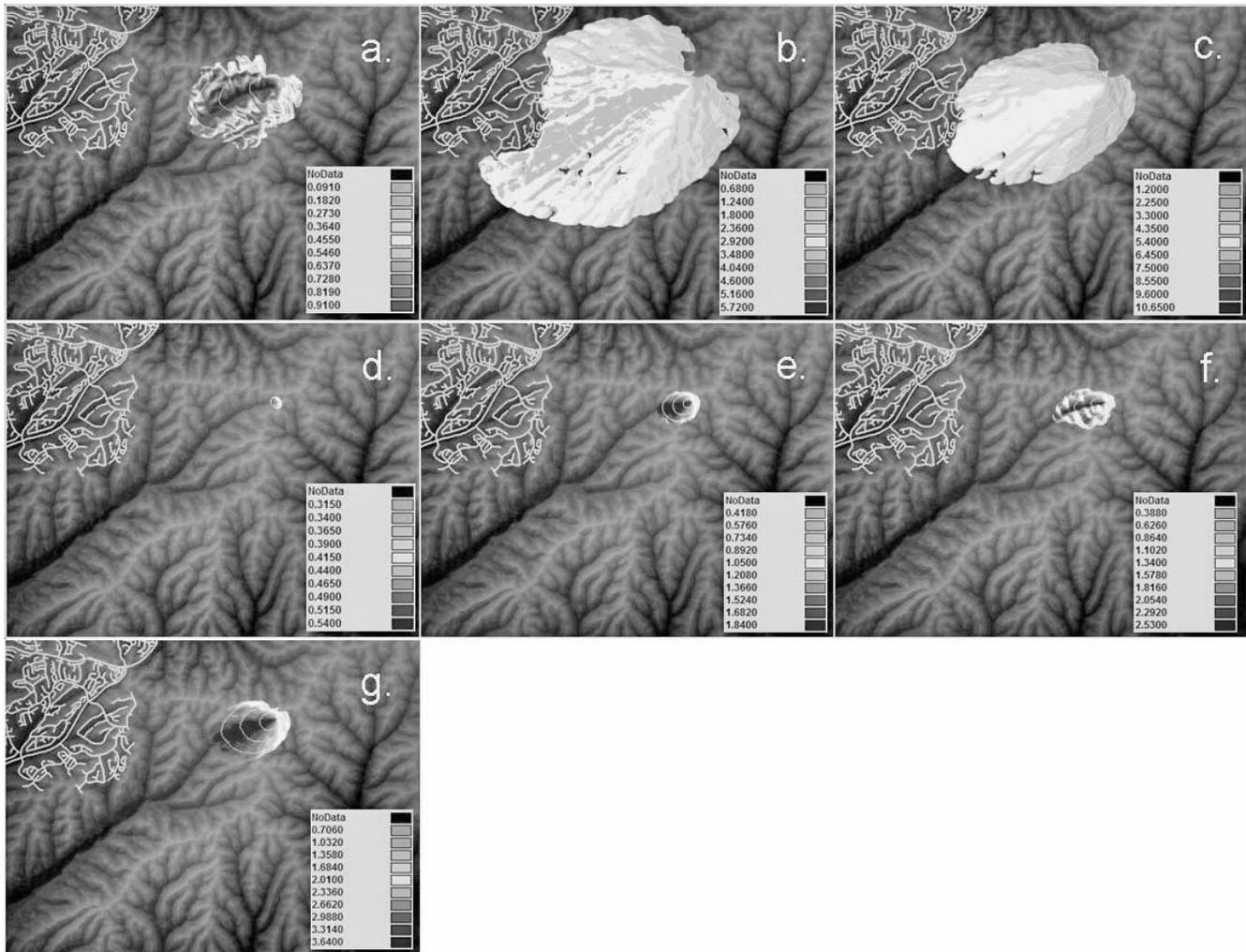


Fig. 2. Simulated fire spread for 3 h under 50th percentile historic weather conditions (September–October) in (a) foothill grasses, (b) nonnative grasses, (c) scrub oak chaparral, (d) coast live oak, (e) chamise chaparral, (f) southern mixed-chaparral, and (g) Diegan coastal sage scrub community types in San Diego County, California.

100.6 kg/ha of air pollution, sequestered 93,387 kg/ha of carbon, and reduced stormwater runoff by 40.4 m³/ha. For all grassland and shrubland vegetation types, CITYgreen calculated zero benefits.

DISCUSSION

Based on past fires in the region, estimated flame lengths (Table 3) may be low, especially under Santa Ana conditions. Even though fires in most vegetation types reached the subdivision within 3 h, not all fires would necessarily result in loss of homes. For example, because of the relatively lower flame lengths in the foothill grassland community, if the homes had proper noncombustible siding materials, they would likely survive direct frontal impingement of the fire, even under Santa Ana conditions. However, it must be noted that not all homes in a wildland fire are consumed by direct flame contact. Many structures are ignited via lofted embers, which land either on combustible roofs and decks or enter through exposed vents or windows (Cohen 2000). Thus, burning embers

from a fire in the coast live oak vegetation type could potentially ignite homes even though the fire never reaches the subdivision. Further, some homes could have tremendous clearance and be built with flame-resistant construction materials, yet still be at high risk because of location in a topographically susceptible area such as the top of a slope or in a chimney. Thus, pre-fire management in the wildland–urban interface must not solely be based on the reduction of fuels, but also must contain elements of home construction, home placement, and landscaping, such that homes can withstand a wildfire in the absence of any suppression actions, which occurred in the early, chaotic stages of the 2003 Fire Siege.

Of note, one of the more explosive vegetation communities in terms of both rate of spread and fire intensity was the nonnative grass community, which should therefore be largely avoided adjacent to human development. However, an increasing population regularly leads to more potential ignition sources and subsequently greater fire frequency, which has been shown to cause a conversion away from native chap-

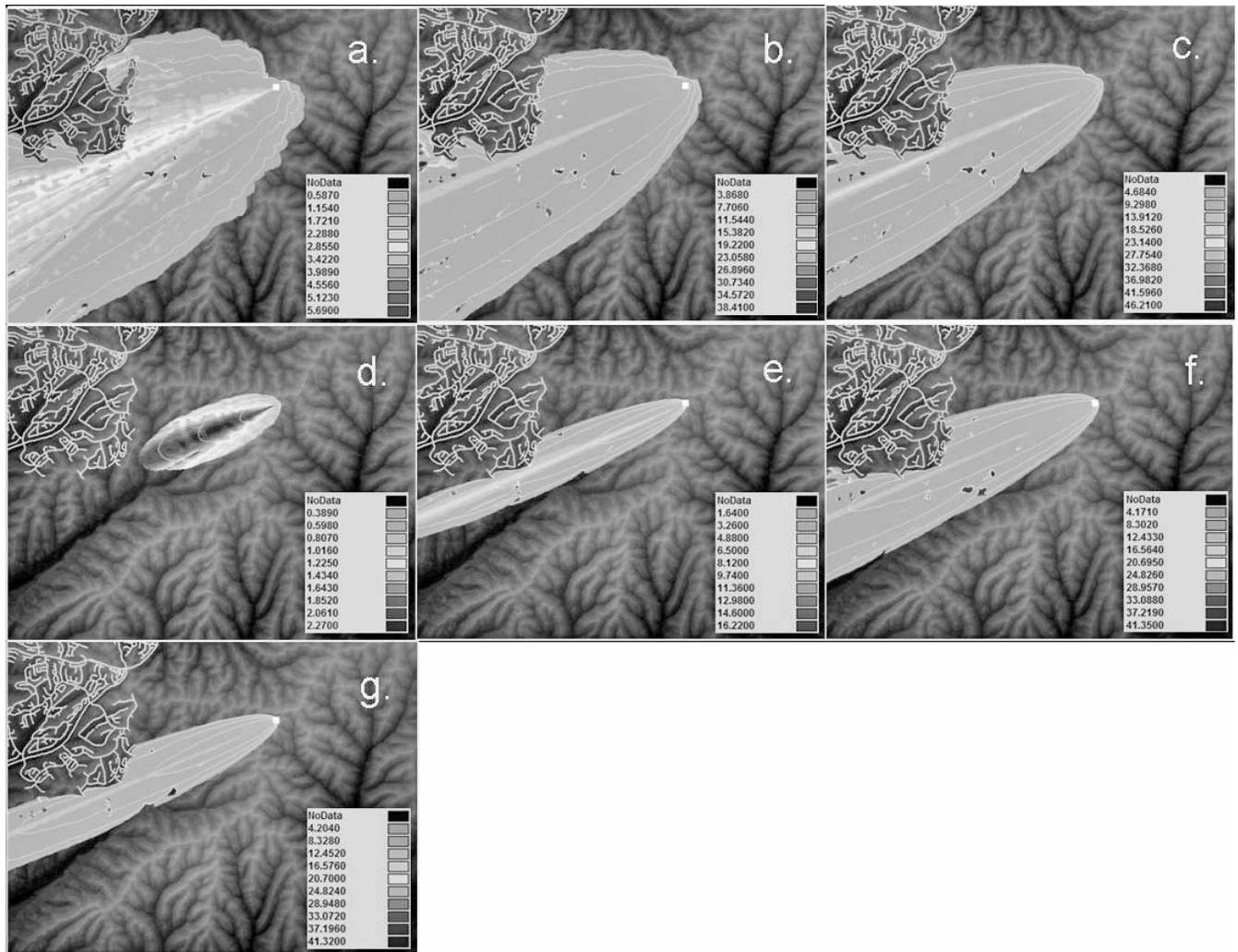


Fig. 3. Simulated fire spread for 3 h under 97th percentile historic weather conditions (September–October) in (a) foothill grasses, (b) nonnative grasses, (c) scrub oak chaparral, (d) coast live oak, (e) chamise chaparral, (f) southern mixed-chaparral, and (g) Diegan coastal sage scrub community types in San Diego County, California.

arral to nonnative grasses (Keeley 2001). Thus, expanding developments in southern California face a self-perpetuating fire and ecological dilemma.

Obviously, one of the more unexpected results of the analysis was the absence of any quantifiable benefits for grassland and shrubland vegetation types. American Forests developed CITYgreen with models based largely on landscapes in the eastern United States and exclusively with trees. For example, calculations of carbon sequestration are based exclusively on tree canopy cover (Nowak and Rowntree 1991, McPherson et al. 1994). And calculations of air pollution removal are based, in part, on pollution data from 10 cities in the United States, the nearest of which to San Diego are Denver, Colorado, and Seattle, Washington. American Forests markets CITYgreen as “calculating the value of nature” and has successfully performed urban ecosystem analyses (UEAs) in eastern cities such as Atlanta, Georgia, Roanoke, Virginia, and Charlotte, North Carolina. In their recent UEA of San Diego (American Forests 2003), they demonstrated not only landscape changes in vegetation cover types, but

also monetary savings that tree canopy cover provided and the monetary losses San Diego has experienced due to the loss of tree canopy cover. However, because the dominant cover types in the area are shrublands and grasslands, benefits derived from trees are only a part of the story there. Thus, in areas of the western United States where trees are not historically the major vegetation type, land managers and policymakers must recognize the limitations of CITYgreen and use caution in the interpretation of its results.

Because CITYgreen did not quantify benefits for any of the grassland or shrubland community types, it was impossible to adequately evaluate the different vegetation communities in terms of the best mix of benefits and fire risk. Because coast live oak showed relatively benign fire behavior and demonstrated tangible benefits, it would seem to be the best option in the area. However, the xeric, Mediterranean climate of San Diego constrains coast live oaks largely to canyon bottoms and occasionally to moister north-facing slopes. Even if there was a hypothetical ability to mass-irrigate the landscapes there to facilitate coast

live oak, a dilemma in the area is that large trees are commonly cut down by homeowners because they are perceived to degrade residential views.

MANAGEMENT IMPLICATIONS

Even though simulated fire behavior varied considerably by vegetation type in our study, few developments in the wildland–urban interface of southern California should be considered fire “safe” during extreme conditions. Vegetation types and structures, construction design, infrastructure, and suppression capabilities all play a role in determining the risk to a given structure. Thus, vegetation management cannot be relied on alone to reduce the risk of wildfire to human developments. Further, fire managers should be aware that although the new 30.48-m (100-ft) clearance regulation is intended to reduce fire intensity, there may be potential for unexpected consequences that could unintentionally increase fire behavior (Dicus and Anderson 2005). For example, thinning some eucalyptus (*Eucalyptus* spp.) stands in San Diego has been shown to increase the amount of nonnative grasses and shrubs in the understory, which created an even more explosive situation than before the thinning occurred (M. Scott, Rancho Santa Fe Fire District, personal communication).

Even though we have reported no benefits, native grasslands and brushlands provide many societal and ecological benefits. Therefore, fire managers must understand the potential benefits of vegetation in an area and recognize that fire risk can be significantly reduced without totally denuding the landscape of vegetation. There is a need in San Diego and throughout southern California for multidisciplinary collaboration to reduce the cycle of repetitive loss from wildfires while simultaneously maximizing other societal values. Laudably, many interface areas in California have initiated local “FireSafe Councils,” which target diverse stakeholders such as fire personnel, landscapers, insurance agents, environmentalists, and academics to seek tangible ways to reduce the fire risk locally.

Land managers and policymakers should use caution when utilizing CITYgreen and understand its limitations. Although a UEA readily shows changes in land use and vegetation, its inability to include non-tree vegetation in its analyses of societal benefits should cause users a certain degree of skepticism in areas such as San Diego, where trees are not the dominant vegetative cover type.

CONCLUSIONS

Vegetation community types will vary in fire behavior and, although not shown here, provide various levels of benefits based on the species and structure of the vegetation on the landscape. Thus, fire managers in the wildland–urban interface should not utilize a one-size-fits-all, clear-at-all-costs mentality in fuels

management. To ensure sustainable developments in the wildland–urban interface, stakeholders from a diversity of disciplines and worldviews must collaborate to determine the best management plan for a given area that simultaneously reduces fire risk and also maximizes the benefits that different vegetation communities provide.

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COMPARISON OF LIVE FUEL MOISTURE SAMPLING METHODS FOR BIG SAGEBRUSH IN UTAH

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ABSTRACT

Live fuel moisture (LFM) has been identified by managers as well as the scientific literature as a key driver of fire behavior in fuel types dominated by live vegetation (i.e., shrublands). Recognizing this, fire managers in Utah use LFM values as a factor in making both tactical and strategic decisions in fire suppression and prescribed burning. Current methods to quantify LFM through field sampling have been based on publications developed for fuel types that are very different from those found in the Great Basin. No research has been conducted regarding the most appropriate procedures to follow to obtain accurate and comparable LFM results for sagebrush fuel types. This proposed research aims to improve local knowledge by determining the most appropriate sampling procedures for assessing LFM in Utah's big sagebrush (*Artemisia tridentata*). This will involve analyzing how variability in data collection affects LFM values and recommending a methodology. Sampling methods suggested in the literature as well as those most commonly utilized by field technicians will be compared and statistically analyzed to detect significant differences in resultant LFM values. In addition, final values from each method will be run through the National Fire Danger Rating System to determine if there are differences in fire danger indices and staffing levels using various collection methods. By developing scientifically sound, standardized methods, fire managers will benefit from improved accuracy and comparability of LFM values.

Citation: Brown, A., P.N. Omi, and J. Pollet. 2007. Comparison of live fuel moisture sampling methods for big sagebrush in Utah [abstract]. Page 221 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

MONITORING FIRE EFFECTS IN GRASSLANDS ON NATIONAL PARK UNITS OF THE NORTHERN GREAT PLAINS

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ABSTRACT

The Northern Great Plains (NGP) Fire Ecology Program began installing fire effects monitoring plots in national park units of the NGP in 1997. Since then, >200 plots have been installed in forest stands, shrublands, grasslands, and riparian areas of ten park units. The general goal of the NGP Fire Ecology Program is to provide vegetation information to the fire management and resource management programs of these parks. This presentation will focus on 41 grassland plots (27 native prairie and 14 nonnative grass) in six park units that have been sampled at least four growing seasons following application of prescribed fire. Prescribed fire has been applied to 28 of these plots once and to 13 plots twice. Park units included in this analysis are Badlands National Park, Devils Tower National Monument, Knife River Indian Villages National Historic Site, Scotts Bluff National Monument, Theodore Roosevelt National Park, and Wind Cave National Park. Plot sampling protocols used are standard National Park Service (NPS) protocols outlined in the NPS *Fire Monitoring Handbook*. Percent cover data were collected using point-intercept method along one 30-m transect at each site. Data will be analyzed to compare pre-burn and 4- or 5-y post-burn values to detect changes in percent cover and relative cover for individual species and for functional guilds of species. The most common goals of prescribed fire in the NGP parks are to maintain native prairie and reduce nonnative grass species. We will also assess the success at which prescribed fire has been used to meet these goals.

Citation: Thorstenson, A., and C.L. Wienk. 2007. Monitoring fire effects in grasslands on national park units of the northern Great Plains [abstract]. Page 222 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

ANALYSIS OF FUEL VARIABILITY WITHIN THE ROCKY MOUNTAIN REGION: INTEGRATION OF FIELD DATA, GEOSPATIAL INFORMATION, AND SPATIAL STATISTICS

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ABSTRACT

The integration of spatial information (remotely sensed data, geographic information system [GIS], Global Positioning System [GPS]), and spatial statistics are effective for modeling coarse-scale and fine-scale ecological variability and for the prediction of fuel loading, variability, and vegetation characteristics. We proposed new techniques to conduct fuel-vegetation surveys based on pixel nested sampling (20×20 -m) designs at different landscape-scale levels at Grand Teton National Park (GRTE)–Bridger Teton National Forest (BTNF), Wyoming, and (15×15 -m) at Rocky Mountain National Park (ROMO), Colorado. Through geospatial statistical modeling and mapping, fuel loadings will be forecast across the landscape. To predict the fuel parameters and forest characteristics at GRTE–BTNF, we will use a new geospatial statistics model using spatial autocorrelation and cross-correlation statistics, trend-surface analysis, and stepwise regression. This process is based on the ordinary least squares or spatial autoregressive, generalized least squares estimates, and generalized linear models. Field data, environmental characteristics, remote sensing, and GIS data will be integrated with spatial statistics to estimate coarse-scale variability in vegetation, fuel parameters, and forest characteristics. Modeling of the spatial continuity of fine-scale variability will be based on binary regression classification trees, kriging, and co-kriging. Semi-variogram models will be selected for the lowest values of corrected Akaike Information Criterion statistics when kriging is used. Using this method, we hope to define a new protocol for fuel modeling and mapping within GRTE–BTNF. The new approach will also provide a cost-effective tool for identifying areas currently affected or vulnerable to invasion by exotic species, as well as assist with other issues of landscape management (i.e., forest fuel loading, wildfire in relation to weeds occupying the landscape, and other factors of concern to resource management teams at GRTE–BTNF and ROMO). This research also provides a growing body of knowledge upon which park managers may base decisions that best preserve the park's natural richness while allowing its beauty to be enjoyed.

Citation: Kalkhan, M.A., and K.E. Brown. 2007. Analysis of fuel variability within the Rocky Mountain region: integration of field data, geospatial information, and spatial statistics [abstract]. Page 223 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

NEW TOOLS FOR ASSESSING LANDSCAPE SCALE OF VEGETATION AND WILDFIRE HAZARDS: A CASE STUDY FROM THE ROCKY MOUNTAIN REGIONS

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ABSTRACT

Integration of spatial information (i.e., remote sensing, geographic information system [GIS], and Global Positioning System [GPS]) with geospatial statistics can be used to investigate the spatial relationships among plant diversity, fuels, wildfire severity, and post-fire invasion by exotic plant species through linkage of multiphase sampling design and multiscale nested sampling field plots (pre- and post-fire). This technique provides useful information and new tools for describing and forecasting landscape-scale fire regimes, invasive plants, and ecological and environmental characteristics for the Cerro Grande Fire site, Los Alamos, New Mexico; Hayman Fire and High Meadow sites, Colorado; and other areas within the Rocky Mountain regions. Results of trend-surface models that describe the coarse-scale spatial variability to predict the distribution, presence, and patterns of native and exotic species, and fuel loading (tons per hectare) using stepwise multiple regressions based on the ordinary least squares (OLS) and spatial autoregressive (SAR) methods will be presented. Models with small variance were selected. In addition, the residuals from the trend-surface model based on the OLS or SAR procedure were modeled using binary regression trees and ordinary kriging for modeling small-scale variability based on semi-variogram models. All models were selected based on lowest values of standard errors, corrected Akaike Information Criterion statistics, and high R^2 .

Citation: Kalkhan, M.A. 2007. New tools for assessing landscape scale of vegetation and wildfire hazards: a case study from the Rocky Mountain regions [abstract]. Page 224 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

ECOLOGICAL DIVERSITY IN CHAPARRAL FOLLOWING PRESCRIBED FIRE AND MASTICATION TREATMENTS

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ABSTRACT

Chaparral fire management has become increasingly challenging as California's population expands into the wildland fringes. Fuel management techniques such as out-of-season prescribed fire and mastication (mechanical shredding) are often used to reduce wildfire risk in chaparral; however, the ecological consequences of these treatments have not been carefully studied. To address this issue, we are investigating the effects of fall, winter, and spring prescribed fire and mastication in northern California chaparral. We are targeting three questions with our research: 1) Do plant and bird species composition–abundance differ between prescribed fire and mastication? 2) Do plant and bird species composition–abundance vary with treatment season (fall, winter, spring)? 3) Does deer herbivory alter plant species composition–cover–abundance? Pre- and post-treatment monitoring of plant and bird recovery has been ongoing since 2001 and will continue through 2005. Final data analysis will be available in 2006. With >100 acres of replicated experimental treatments, this research project is one of the only fire management studies of its kind. Preliminary results suggest that plant and bird responses vary significantly after these fuel reduction treatments, particularly in relation to nonnative grass cover and bird abundance. In particular, we are noticing substantially higher abundance and cover of nonnative plant species in all mastication plots. Nonnative grasses are especially prevalent and will not only dominate resources at the present time, but they will have the advantage of depositing seeds in the soil for future reestablishment. Nonnative grasses also have the potential to increase the ecosystem's flammability because they cure earlier than native grasses and have more extensive cover. We are also noticing substantial trends in bird composition and abundance. To date, we have found 40 bird species and 800+ individuals using post-fire habitat compared to 14 species and 60+ individuals in post-mastication habitat. Furthermore, fire and mastication plots do not share the same common species. For example, the most common post-fire species are western scrub jay (*Aphelocoma californica*), spotted towhee (*Pipilo maculatus*), and Bewick's wren (*Thryomanes bewickii*). The most common post-mastication species are California quail (*Callipepla californica*) and dark-eyed junco (*Junco hyemalis*). In effect, current chaparral fuel management practices may be significantly altering floral and faunal community composition. These patterns have not been well studied in northern California but clearly need research, especially as fuel reduction becomes more extensive in order to protect a growing number of homes from wildfire. Existing chaparral fire studies have often been opportunistic and inadequately replicated to take advantage of post-wildfire conditions. In addition, few comparative studies have focused on fire versus mastication and their seasonal timing. As a result, this research project aims to provide robust, region-specific information to chaparral managers in order to improve the quality of future fuels management.

Citation: Potts, J., and S. Stephens. 2007. Ecological diversity in chaparral following prescribed fire and mastication treatments [abstract]. Page 225 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

FIRE EFFECTS ON THE STRUCTURE OF WOODY VEGETATION OF *CERRADO DENSO*, A SAVANNA-LIKE VEGETATION OF CENTRAL BRAZIL

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ABSTRACT

The cerrado, a savanna-like vegetation, covers 25% of the Brazilian territory. *Cerrado denso* is the denser form of cerrado, with trees reaching 8 m in height and an herbaceous layer composed of sparse herbs and grasses that dry out from April to September, when fires are common in the region. In 1992, at the Instituto Brasileiro de Geografia e Estatística Ecological Reserve, 26 km south of Brasília, Brazil, three 10-ha plots, protected from fire for 18 y, were selected to investigate the effects of fire on the mortality rate of the *cerrado denso* woody vegetation. The fire regime selected was biennial fires at the beginning (Jun), middle (Aug), and end of the dry season (Sep). Before the fires, there were 2,040 (Jun), 2,188 (Aug), and 1,870 (Sep) live individuals/ha in the plots, distributed among 59, 62, and 68 species. The estimated aboveground tree biomass was 20.7 t/ha (Jun), 35.3 t/ha (Aug), and 38.7 t/ha (Sep). In the herbaceous layer, the biomass was 7.5 t/ha in the June and August plots and 5.4 t/ha in the September plot. In 2001, after five biennial prescribed fires, the cumulative mortality was 32% in the June plot, 34% in the August plot, and 41% in the September plot. These numbers increased to 48% (Jun), 54% (Aug), and 74% (Sep) when the numbers of stems destroyed (dead + topkill) are considered. The fires resulted in a loss of approximately 26% of the woody aboveground biomass in the June plot, approximately 40% in the August plot, and approximately 45% in the September plot. There was an increase of 1.8 t/ha in the biomass of the herbaceous layer in the June plot, 1.0 t/ha in the August plot, and 4.9 t/ha in the September plot. The results suggest that the structure of the woody vegetation in the *cerrado denso* is highly affected by high fire frequency. The reduction in the number of live stems and the increase in the herbaceous layer biomass may result in changes in the energy balance of the system, with significant alterations in the carbon and water fluxes.

Citation: Sato, M.N., H.S. Miranda, and P.J. Riggan. 2007. Fire effects on the structure of woody vegetation of *cerrado denso*, a savanna-like vegetation of central Brazil. Page 226 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

NONNATIVE INVASIVE PLANTS AND FIRE: LITERATURE REVIEWS AND KNOWLEDGE GAPS

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ABSTRACT

The Fire Effects Information System (FEIS) provides reviews of the scientific literature regarding basic biology and fire ecology of nearly 1,000 plant and animal species. Managers use FEIS in planning for post-fire rehabilitation, use of fire in ecosystem maintenance and restoration, and other management. Scientists and members of the public also use the system. Reviews of 60 nonnative invasive plant species have recently been revised in or added to FEIS (<http://www.fs.fed.us/database/feis/plants/weed/weedpage.html>). Most of these species occur in grassland or shrubland ecosystems. As reviews were written, gaps in science-based knowledge about the relationships between fire and these 60 nonnative invasive species were systematically identified. Fewer than half of the species studied had information on heat tolerance, post-fire establishment, effects of varying fire regimes (severities, seasons, and intervals between burns), or long-term effects of fire. Information was generally available, although sometimes incomplete, on biological and ecological characteristics relating to fire. Most information about species distribution used too coarse a scale or nonsystematic observations, rendering it of little help in assessing invasiveness and susceptibility of ecosystems to invasion, especially in regard to fire. Quantitative information on nonnative plants' impacts on native plant communities and long-term effects on ecosystems was sparse. Researchers can improve the knowledge available to managers by applying rigorous scientific methods and reporting the scope of the research, in both scientific papers and literature reviews. Managers can use this knowledge most effectively by applying scientific findings with caution appropriate to the scope of the research, monitoring treatment results over the long term, and adapting management techniques according to what is learned.

Citation: Munger, G.T., K. Zouhar, and J.K. Smith. 2007. Nonnative invasive plants and fire: literature reviews and knowledge gaps [abstract]. Page 227 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

MODIFYING NATIONAL PARK SERVICE FIRE EFFECTS SAMPLING PROTOCOLS TO MEET THE NEEDS OF THE FIRE ECOLOGY, INVENTORY AND MONITORING, AND EXOTIC PLANT MANAGEMENT PROGRAMS

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ABSTRACT

In 1997, the Northern Great Plains Fire Effects team began monitoring the effects of prescribed fire activities on vegetation in ten National Park Service (NPS) units as part of the NPS's national Fire Effects Program. In 1999, as a result of the Natural Resource Challenge federal legislation, the National Park Service established two new national programs interested in monitoring vegetation. The Exotic Plant Management Program is organized into regional teams (EPMTs) that map the extent of and treat invasive plant species, while the Inventory and Monitoring Program is organized into networks charged with tracking the ecosystem health of NPS units by monitoring indicative "vital signs." In the northern Great Plains, the geographic overlap and mutual interests of these three programs has led to a cooperative effort to design a vegetation-monitoring protocol that will meet the needs of all three programs in ways that the individual programs' monitoring alone could not. We present the results of this effort so far. We focus on 1) the development of monitoring objectives, 2) overcoming issues in sampling design across space and through time, and 3) the results of a pilot study designed to compare the efficiency and efficacy of two methods of measuring plant community composition in the major vegetation types that will be monitored in this region.

Citation: Wienk, C.L., A.J. Symstad, A. Thorstenson, D. Licht, and C.W. Prosser. 2007. Modifying National Park Service fire effects sampling protocols to meet the needs of the Fire Ecology, Inventory and Monitoring, and Exotic Plant Management programs [abstract]. Page 228 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

THE PUU ANAHULU WILDFIRE MANAGEMENT STUDY: DEVELOPMENT OF ROADSIDE FUELS REDUCTION TECHNIQUES FOR LEEWARD HAWAI‘I

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ABSTRACT

Invasion of native lowland dry forests and shrublands on the island of Hawai‘i by fountaingrass (*Pennisetum setaceum*) has created a fine fuel bed capable of carrying surface fires at frequent, regular intervals. A corresponding expansion of the human population into arid leeward areas has resulted in an increase in roadside ignitions. These two factors are contributing to the conversion of native dry forest and shrubland to savanna and grassland, and have resulted in dramatic declines or extinction of >30 endemic plant and animal species. The new fire regime also increasingly threatens life and property throughout the northwest region of the island. In an effort to develop and refine roadside fuels management techniques that could break this cycle, we evaluated the effects of single applications of prescribed burning, cattle grazing, and aerially applied herbicide, applied alone and in combinations between January and May 2004. Fuel loading, vegetation response, and soil seedbank composition were monitored over a 1-y period. Prescribed burns were the first burns conducted on state land and entailed extensive cooperative planning and coordination by local state, county, and federal land management and civil service agencies. Burns caused an immediate reduction in fountaingrass fuel load. Cattle grazing was not effective in reducing fuel loading in this study due to low stocking rates applied. Aerially applied glyphosate herbicide effectively killed the grass and initiated a process of decomposition. Glyphosate treatments resulted in a release of the existing and predominantly nonnative soil seedbank. The combined burn × graze × glyphosate treatment resulted in the greatest increase in plant species richness (mean = 5).

Citation: Castillo, J.M., M. Nakahara, D. Weise, R. Vihnanek, G. Enriques, A. McAdams, L. Ford, R. Moraga, B. Babb, M. Thorne, E. Nonner, and D. Frohlich. 2007. The Puu Anahulu wildfire management study: development of roadside fuels reduction techniques for leeward Hawai‘i [abstract]. Page 229 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

EFFECTS OF CATTLE GRAZING, GLYPHOSATE, AND PRESCRIBED BURNING ON FOUNTAINGRASS FUEL LOADING IN HAWAI'I

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ABSTRACT

Crimson fountaingrass (*Pennisetum setaceum*) is a nonnative invasive grass that has occupied a significant portion of the western side of the island of Hawai'i. As a result, several fires in excess of 4,049 ha have occurred in the area over the past 20 y. We are studying the effectiveness of cattle grazing, aerial application of glyphosate herbicide, and prescribed burning to reduce the fuel loading of fountaingrass in this dry, tropical setting. Grazing and prescribed burning were applied as whole-plot treatments to plots ranging in area from 2.4 to 6.5 ha; glyphosate herbicide was aerially applied as a split-plot treatment. Plots were burned in January and February 2004, aerially sprayed in March and May 2004, and grazed by cattle in March and April 2004. Fuels were sampled prior to treatment and at periodic intervals over the first year following treatment. Fuel and fire behavior variables were measured during the prescribed burns. Pre-treatment fuel loading ranged from 9 to 11 Mg/ha and fuel height averaged 0.5 m. Observed dead fuel moistures were 12–16% and live grass fuel moisture was >270% during the prescribed burns. Fire spread rates up to 16.8 m/min and flame heights up to 3.7 m were observed. Prescribed burning and glyphosate reduced fuel loads by 46% and 14%, respectively. Fuel height was reduced by 8–48%. Grazing did not have a significant effect in this study. Fuel height did recover and fuel loads did not recover to pre-treatment levels within 13 mo of the prescribed burns.

keywords: crimson fountaingrass, fire, flame length, fuels reduction, glyphosate, grazing, Hawai'i, leeward, *Pennisetum setaceum*, rate of spread.

Citation: Castillo, J.M., G. Enriques, M. Nakahara, D. Weise, L. Ford, R. Moraga, and R. Vihnanek. 2007. Effects of cattle grazing, glyphosate, and prescribed burning on fountaingrass fuel loading in Hawai'i. Pages 230–239 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

INTRODUCTION

In the tropics, grass invasions have been observed to cause increased fire frequency (McDonald et al. 1988, D'Antonio and Vitousek 1992, Smith and Tunison 1992). Grass invasions into dry vegetation communities of Hawai'i have resulted in recurrent wildfires that reduce woody plant cover and simplify plant

community structure (Cuddihy and Stone 1990, Hughes et al. 1991, D'Antonio and Vitousek 1992, Castillo 1997). Resultant wildfires provide for increased solar radiation, higher near-ground wind speeds, and lower fine fuel moisture (Blackmore and Vitousek 2000). These conditions of higher fine fuel loads, lower fine fuel moisture, and higher near-ground wind speeds are conducive to fire ignition and spread (Freifelder et al. 1998).

Crimson fountaingrass (*Pennisetum setaceum*) was

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introduced into Hawai'i at the turn of the last century and now occupies >80,000 ha on the dry, leeward side of the island of Hawai'i. This grass invasion altered the surface fuel structure, which has increased the occurrence of large wildfires: five class G fires up to 4,000 ha have occurred in this area since the 1980s. Fountaingrass has also been introduced in the states of California, Arizona, New Mexico, Colorado, Tennessee, Florida, and Louisiana (USDA 2005). This introduction, along with several other invasive grass species, has resulted in fire spread into areas where fire occurrence was previously rare (Brooks and Pyke 2001).

Wildland fires ignited along road corridors present a threat to adjoining vegetation communities. Several herbicides are currently registered in Hawai'i for fountaingrass control; the active ingredients are diuron, hexazinone, imazapyr, and isopropylamine salt of glyphosate (National Pesticide Information Retrieval System 2006). Glyphosate (sold as Roundup®) has been used on the western side of Hawai'i to control fountaingrass (Motooka 2000, Cabin et al. 2002). In a study that evaluated the duration of cover reduction by various herbicides, a low rate of glyphosate applied to fountaingrass 3 mo after a wildfire reduced cover by 75% 2 y following treatment. The low rate of glyphosate was as effective as more intensive treatments such as herbicide and hand-pulling (Castillo 1997). Tunison (1992) reported that fountaingrass could be reduced in Hawai'i Volcanoes National Park with a great deal of labor by manually removing clumps.

There is some disagreement about the importance of fire in the evolution of Hawaiian biota (Vogl 1969, Loope 2000); however, it is well accepted that wildfire has deleterious effects on Hawaiian ecosystems. Prescribed fire was not recommended as a tool to eradicate fountaingrass (D'Antonio and Vitousek 1992); however, it is an effective tool to reduce fuel loading. The use and ecological impacts of cattle grazing and range management in Hawai'i are also well documented (Cuddihy and Stone 1990, Maly and Wilcox 2000). However, cattle reduce grass fuel loading if the grasses are palatable and could possibly be a commercially viable tool to reduce fuel loads in degraded areas dominated by introduced nonnative grasses. We established demonstration areas in western Hawai'i to illustrate the effectiveness and duration of these potential fuel treatments to reduce fountaingrass fuel loading. Because the primary fuel in the study area was fountaingrass and the intent of the project was to demonstrate the effectiveness of potential fuel treatments, large-scale treatments that might be applied operationally to grasslands were considered—prescribed fire, herbicides, and grazing. We present first-year results of this study to reduce the fuel loading of fountaingrass on the leeward side of Hawai'i.

STUDY AREA AND METHODS

Site Description

The study site was located in an extensive stand of fountaingrass located on the western side of the

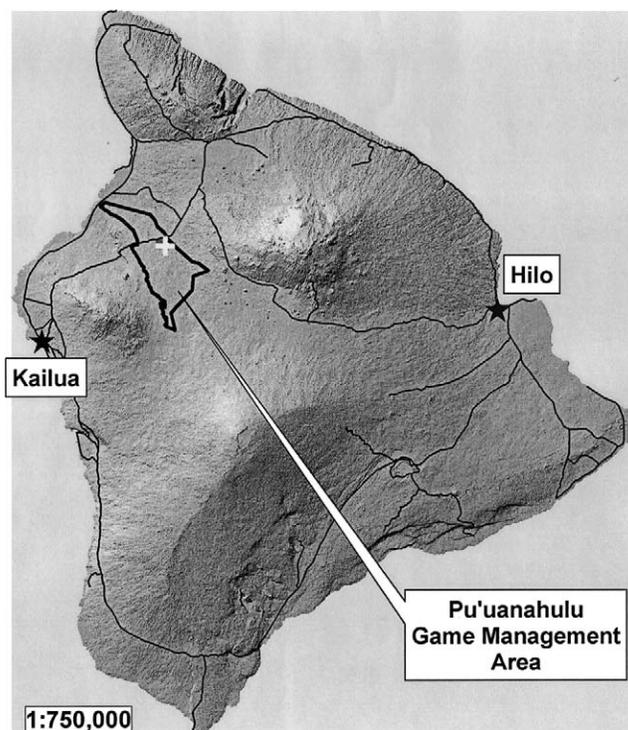


Fig. 1. A "+" identifies the approximate location of Pu'uana'hulu Fuel Treatment Demonstration Area within the Puu An'ahulu Game Management Area, Hawai'i.

island of Hawai'i in the Pu'uana'hulu Game Management Area (Figure 1). The vegetation in the game management area was very degraded relative to native plant populations. Lowland dry forest (Gagne and Cuddihy 1999) likely occupied the site prior to invasion by fountaingrass over the past 90 y. Remnant individual trees from the Hawaiian native dry forest are found within 4 km of the site. Average elevation of the site is 770 m, with a predominant west-facing aspect. The site is situated between two lava flows from Mauna Loa—the 1859 flow and the Ke'amuku flow. The soils are histosols over lava with andisols (Gavenda et al. 1998). The average slope varies from 0 to 15% over most of the site.

A remote automated weather station (RAWS) was established in April 2003 to record weather conditions at the site because the closest weather station was the Kailua-Kona Airport, which is located at sea level approximately 30 km southwest of the site. The monthly average temperatures and relative humidity for April 2003–August 2006 illustrate the relative stability of the climate (Table 1). Precipitation was recorded at the site for various times: 22.7 cm for 14 April 2003 to 28 January 2004, 44.1 cm for 19 April 2004 to 18 April 2005, and 33.0 cm from 19 April 2005 to 18 April 2006. This is similar to the average annual rainfall of 25–50 cm reported by Giambelluca and Schroeder (1998).

Table 1. Monthly summary statistics for Pu'uuanahulu weather station, Hawai'i, from April 2003 to April 2006.

Month	Temperature (°C)			Relative humidity (%)		Solar intensity (W m ⁻²)	
	Mean	Min.	Max.	Mean	Min.	Max.	Mean
Jan	17.8	13.3	23.7	83.8	60.2	712	165
Feb	17.2	12.5	23.3	81.5	58.4	766	192
Mar	18.1	14.0	23.5	87.4	65.3	739	176
Apr	18.7	13.8	24.4	78.9	55.9	872	225
May	20.1	15.3	25.5	80.1	57.7	867	224
Jun	20.5	15.6	26.1	79.4	57.1	826	219
Jul	21.1	16.2	26.7	78.0	55.3	881	235
Aug	21.6	16.6	27.6	77.9	55.2	908	233
Sep	21.4	16.4	27.3	77.5	54.7	888	225
Oct	20.5	15.8	26.4	79.3	55.1	784	192
Nov	19.6	15.0	25.3	81.5	58.7	673	160
Dec	18.3	13.7	24.2	79.1	55.0	651	154

Study Design

In order to provide a fuel break along the Malahoa Highway to reduce the risk of fires spreading from the road upslope to native plant reserves located in the U.S. Army Pohakuloa Training Area (Shaw et al. 1997), the study plots were established on the upslope side of the highway. Combinations of prescribed burning—no burning and grazing—no grazing were applied as whole-plot treatments, and glyphosate herbicide was applied as a split-plot treatment in a factorial design with three replications of complete blocks (Figure 2). The treatments were numbered at the split-plot treatment level (1–8). Each block consisted of four whole plots and eight split plots. Locations of the whole plots were restricted to reduce the total area fenced and to minimize the number of prescribed burns conducted. In the idealized design (Figure 2), a total of six plots would have been burned (three blocks × two plots). However, four of the burns were applied to the whole plots in blocks 1 (burns A, B) and 3 (burns C, E), and one burn (D) was used to burn both whole plots in block 2. The total area of the experiment was approximately 35 ha. Table 2 lists the plots and the fuel treatments applied. The repeated-measures linear model describing the idealized experimental setup contained both fixed and random effects (Equation 1). Repeated-measures analysis of variance was used to test the significance of the effects in this model. The probability level associated with significant effects was set to 0.05.

$$y_{ijklm} = \mu + W_i + \beta_j + \gamma_k + \beta_j\gamma_k + \eta_l + \beta_j\eta_l + \gamma_k\eta_l + \beta_j\gamma_k\eta_l + \tau_m + \beta_j\tau_m + \gamma_k\tau_m + \beta_j\gamma_k\tau_m + \eta_l\tau_m + \beta_j\eta_l\tau_m + \gamma_k\eta_l\tau_m + \beta_j\gamma_k\eta_l\tau_m + e_{ijklm}, \quad (1)$$

where W = block, β = burning, γ = grazing, η = herbicide, τ = time effect, and e = error. The block effect, W , was treated as a random effect to account for the restrictions on treatment location. The significance of the burning (β) and grazing (γ) effects was tested using the sum of squares of the plots within each block, and the significance of the glyphosate (η) effect and its interactions was tested using the residual error sum of squares. Means for each treatment effect were

estimated as linear combinations of the least-squares estimates. The significance of the difference between a treatment mean and its control (no treatment) was tested for all treatment combinations using t -tests. The F -statistics associated with the terms in Equation 1 test a different hypothesis—equality of all levels of a treatment combination (i.e., $H_0: \beta_0 = \beta_1 = 0$ or $H_0: \beta_0\gamma_0 = \beta_0\gamma_1 = \beta_1\gamma_0 = \beta_1\gamma_1 = 0$).

The five prescribed burns were conducted in January 2004, grazing was applied in late April and May 2004, and glyphosate herbicide was applied aerially to areas that would not receive grazing in March, and to grazed areas in May 2004. The burn prescription required a 6.1-m wind speed of 8.3–16.7 km/h from the north, which yielded a midflame wind speed of 5–10 km/h, relative humidity of 60–70%, air temperature of 13–20°C, and 1-h dead fuel moisture content of 15–18%. This prescription was derived using the BEHAVE program to define fuel and weather variables based on desired rate of spread or flame length (Andrews 1986). A black line was established on the downwind side, followed by flank ignition and strip headfires to complete the burning.

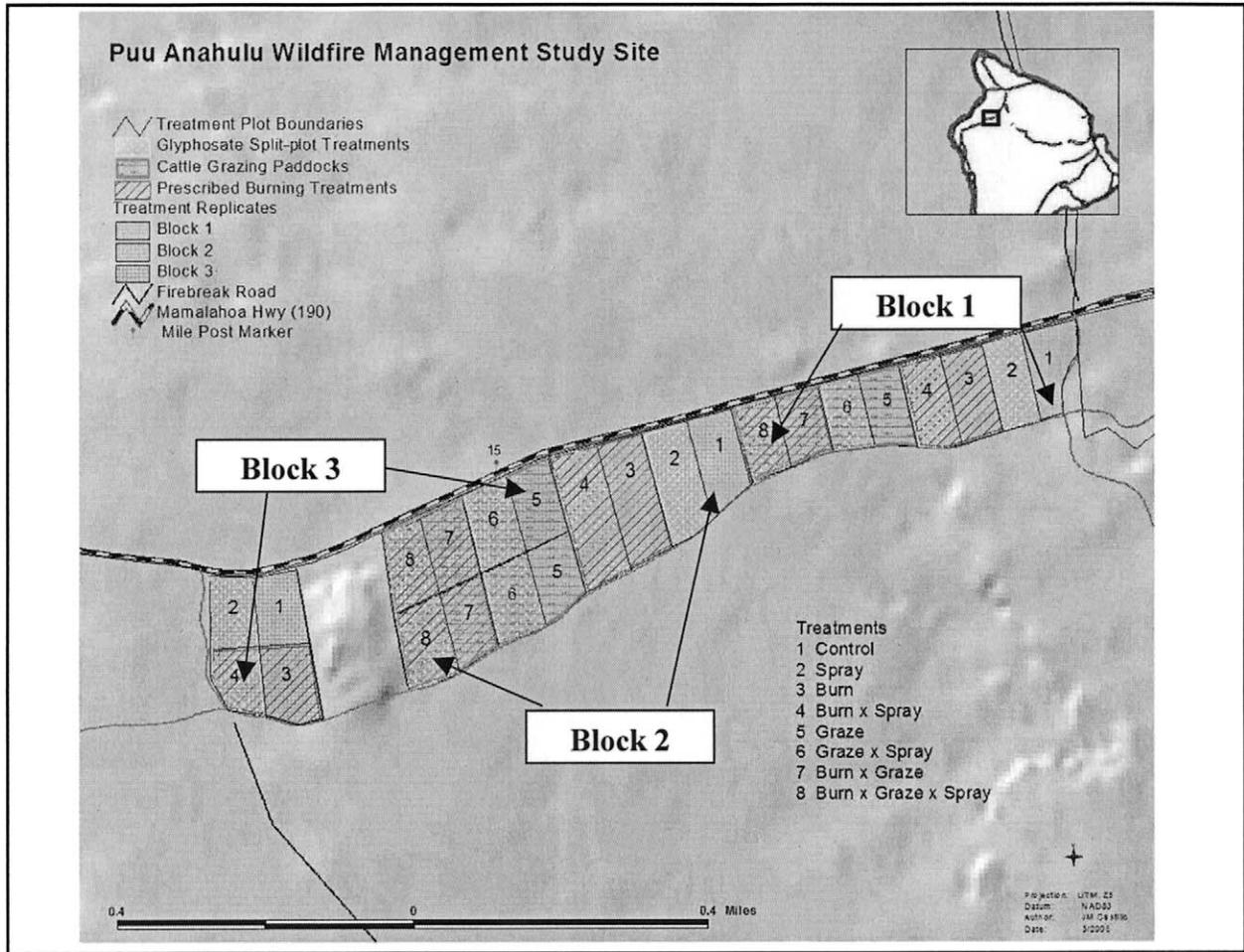
Cattle grazing was applied as a light-intensity pulse graze for a period of 26 d at a rate of 0.24 to 0.32 animal-unit mo. Animals were confined to unburned portions of each plot for the first 6 d and then allowed free choice between burned and unburned areas for the remaining 20 d. Glyphosate was applied at the rate of 2.8 kg active ingredient/ha from a Hughes 500D helicopter equipped with a 9.1-m boom sprayer.

Treatment Fuel Sampling

Average height, absolute cover of all herbaceous material, and fuel loading were measured on 10 randomly located 1-m² samples within a split plot (Ford 2005). Fuel loading samples were collected by clipping all material in the square meter and weighed in the field to determine wet weight. A subsample of the fuel load sample was weighed in the field to determine wet weight, was oven-dried at 70°C to a constant weight, and then was reweighed to determine dry weight. Oven safety concerns prevented drying samples at a higher temperature. Subsample moisture content was determined on an oven-dry basis and used to estimate the sample's dry mass. Each split plot was sampled four times: prior to burning (January 2004), after grazing and herbicide applications (February to May 2004), in August 2004, and in March 2005.

Fire Behavior Measurements

Weather, fuel moisture, rate of spread, and flame length data were collected for each of the six plots that were burned. Weather data were recorded by two automatic weather stations located within 0.5 km north and south of the plots and out of the influence of the smoke. Weather data were also collected manually throughout the burns using a sling psychrometer and handheld anemometer typically found in a belt weather kit. Immediately prior to ignition, fuel moisture samples were collected by clipping the live grass that con-



Spray		Spray		Spray		Spray	
8	7	6	5	4	3	2	1
Burn Graze		No Burn Graze		Burn No Graze		No Burn No Graze	

Idealized Block Layout

Fig. 2. Actual and idealized treatment designs to reduce crimson fountaingrass fuel loading for Pu'uuanahulu Wildfire Management Study, Hawai'i, 2004–2005.

stituted most of the fuel bed and placing the samples in a moisture-tight Nalgene® bottle (Nalge Nunc International, Rochester, NY) for later processing. Fuel moisture content was determined by drying 5- to 10-g subsamples from each bottle in a Computrac® Max 1000 moisture analyzer (Arizona Instrument, Tempe, AZ). Rate of spread was estimated for various segments of uniform fuels by measuring time and dis-

tance. Flame length was recorded using a video camera. A target of known size was placed in the plot to estimate flame length from the imagery.

RESULTS

Mean (± SE) oven-dry fuel loadings of the grass and herbaceous fuels prior to treatment in January

Table 2. Treatments to reduce crimson fountaingrass fuel loading, Pu'uuanahulu Game Management Area, Hawai'i, 2004–2005.

Treatment no.	Prescribed burn	Grazing	Glyphosate
1	No	No	No
2	No	No	Yes
3	Yes	No	No
4	Yes	No	Yes
5	No	Yes	No
6	No	Yes	Yes
7	Yes	Yes	No
8	Yes	Yes	Yes

2004 were 11.9 ± 1.0 , 9.8 ± 1.3 , and 9.5 ± 1.6 Mg/ha for the three blocks ($n = 4$) (Table 3). Initial fuel heights were 0.55 ± 0.02 , 0.47 ± 0.04 , and 0.57 ± 0.04 m. The initial fuel loadings and fuel heights did not differ significantly between blocks. Moisture content of the samples averaged $52 \pm 2\%$ across the three blocks. The fuel moisture samples were a composite of litter, standing live herbaceous material, and dead fountaingrass.

Prescribed Fire Behavior

From 1 September 2003 to 25 January 2004, 21 cm of rain were recorded by the RAWS, resulting in fountaingrass that was vigorous, in flower, and very green in color at the time of the burns. While fuel moistures measured to estimate fuel loading averaged 52%, the fuel moisture of the predominantly live grass was >270% (Table 4). Dead fuel moisture content fell in the 12–16% range during the prescribed burns.

The plots were burned over an 8-d period from 27

January to 4 February 2004. The first plot treated was difficult to burn. Conditions, although in prescription, were marginally sufficient for burning, and a large percentage of the plot had to be ignited (Table 4). Multiple strip headfires were lit on a very close spacing (5–10 m between strips). This plot was the first prescribed burn conducted by the Hawai'i Division of Forestry and Wildlife on Hawai'i, so due caution was taken to minimize escape. The observed spread rates for this plot ranged from 0.7 to 2.3 m/min, and approximately 40% of the fuel was consumed (Table 5). As Division of Forestry and Wildlife personnel's experience increased, more aggressive burning occurred, resulting in greater spread rates up to 16.8 m/min, flame heights of 2.7 to 3.7 m, and fuel consumption up to 90%.

Impact of Treatments on Fuel Loading and Fuel Height

Fuel loadings and heights were estimated following the application of the final treatment (March to May 2004) and 6 and 13 mo after the prescribed burns (August 2004, March 2005) (Table 3). The repeated-measures analysis of variance indicated that only the burn treatment and the burn \times time interaction significantly affected fuel loading (Table 6; significant if $P < 0.05$). The time effect was also significant, indicating that the average loading on the site (average of all plots) differed over the four measurement times. The significant burn \times time interaction indicated that the change in fuel loading over time differed between the

Table 3. Estimated fine fuel loading pre-treatment and for 1 y following fuel treatment application to reduce crimson fountaingrass in the Pu'uuanahulu Game Management Area, Hawai'i, 2004–2005.

Treatment				Fuel loading (Mg/ha) ^a							
				Jan 2004		May 2004		Aug 2004		Mar 2005	
Burn	Grazing	Herbicide	Block	Mean	SE	Mean	SE	Mean	SE	Mean	SE
N	N	N	1	9.8	1.3	10.9	0.9	16.1	1.1	12.5	1.0
			2	6.2	2.5	10.1	2.2	12.8	1.9	6.3	1.6
			3	8.0	1.6	6.5	2.0	6.8	0.9	4.8	1.0
N	N	Y	1	12.3	1.3	11.3	2.0	12.0	2.3	7.1	1.1
			2	4.4	1.8	7.9	2.4	9.4	2.2	4.6	1.6
			3	8.4	1.3	7.6	1.5	8.9	2.3	5.9	1.5
Y	N	N	1	14.4	2.6	4.4	1.8	4.7	0.8	4.1	0.7
			2	8.1	3.1	0.5	0.1	2.8	1.2	2.0	0.7
			3	10.5	4.3	1.3	0.3	3.7	0.7	2.7	0.4
Y	N	Y	1	16.2	3.6	1.7	0.5	1.9	0.9	1.7	0.4
			2	13.8	2.6	0.8	0.4	1.8	0.5	1.2	0.2
			3	11.7	2.3	2.2	0.4	1.7	0.2	4.3	0.8
N	Y	N	1	7.4	2.4	9.9	3.1	11.2	2.6	5.1	1.4
			2	9.6	2.3	14.2	2.7	18.9	3.5	6.2	1.2
			3	8.1	2.1	17.0	4.4	10.8	3.2	4.0	1.2
N	Y	Y	1	12.0	2.6	9.6	2.2	12.3	2.7	4.7	1.2
			2	4.4	1.2	6.4	2.1	11.3	2.0	8.7	2.2
			3	3.8	1.2	3.0	1.2	10.6	2.8	5.7	1.6
Y	Y	N	1	10.3	1.8	2.1	1.1	2.4	1.0	3.3	0.8
			2	14.0	2.4	0.7	0.2	1.8	0.3	1.7	0.4
			3	15.5	2.9	1.5	0.3	4.0	0.8	3.5	0.6
Y	Y	Y	1	12.6	2.8	0.5	0.1	1.4	0.7	1.6	0.7
			2	15.2	2.4	0.8	0.2	1.0	0.2	1.9	1.0
			3	12.7	1.8	0.6	0.2	2.0	1.0	1.4	0.5

^a Values are plot mean and within-plot standard error ($n = 10$). 1 Mg/ha = 1 metric ton/ha = 0.45 English ton/acre.

Table 4. Estimated fuel and weather conditions during prescribed burns in crimson fountaingrass near Pu'uana'hulu, Hawai'i, in 2004.

Block	Treatment no. ^a	Date	Time	Status ^b	Moisture content (%)	Temperature (°C)		Relative humidity (%)
						Air	Fuel	
1	3, 4	28 Jan	1050	D	14.4	21.3	26.5	66
1	7, 8	3 Feb	1000	D	15.3	19.5	23.8	66
1	7, 8	3 Feb	1000	L	277.7	19.5	23.8	66
1	7, 8	3 Feb	1240	D	13.7	19.9	20.4	71
2	3, 4	3 Feb	1445	L	275.3	19.8	18.9	74
2	3, 4	3 Feb	1520	D	13.9	19.4	19.1	73
2	3, 4	3 Feb	1645	D	12.7	20.0	19.7	72
2, 3	7, 8	4 Feb	1000	D	16.0	19.7	24.7	71
2, 3	7, 8	4 Feb	1000	L	288.5	19.7	24.7	71
3	3, 4	27 Jan	1130	D	13.5	23.1	30.7	55
3	3, 4	27 Jan	1230	L	288.3	21.2	25.8	74

^a Treatment no.: 3, burned; 4, burned and glyphosate applied; 7, burned and grazed; 8, burned, grazed, and glyphosate applied.

^b Status: D, dead; L, live.

burned and unburned plots (Figure 3a). The *P*-value of the herbicide × time interaction (0.062) was slightly greater than our level of significance, and *P*-values for some of the tests of the higher order interaction terms fell in the range 0.10 > *P* > 0.05. The herbicide × time interaction suggests that the temporal change in fuel loading on the split plots treated with glyphosate was different from the temporal change on the split plots not treated with glyphosate.

The mean fuel loadings estimated by least squares for each treatment and various combinations of treatments ranged from 4.3 to 9.7 Mg/ha (Table 7). Mean fuel loadings for all treatment combinations with prescribed fire were significantly less than the corresponding unburned control mean. The mean loadings for the grazing and glyphosate treatments were not statistically different from the untreated control. Averaged over the entire experiment, prescribed burning reduced fountaingrass fuel loading by 46%, grazing did not reduce fuel loading, and glyphosate reduced fuel loading by 14% (Table 7).

Mean fuel heights for all treatments except grazing alone were significantly less than the corresponding untreated mean heights (Table 7). Mean fuel height reduction ranged from a low of 8% (Table 7: G [grazing]) to a high of 48% (Table 7: BGH [burning, grazing, glyphosate]). When averaged over the entire experiment, prescribed fire reduced fuel height by 23% and glyphosate reduced fuel height by 17%. Species composition data were collected but not presented here. These data have yet to be analyzed; however,

Table 5. Observed fire behavior and visually estimated fuel consumption in prescribed burns in crimson fountaingrass stands near Pu'uana'hulu, Hawai'i, in 2004.

Date	Block	Treatment no. ^a	Spread rate range (m/min)	Flame height range (m)	Fuel consumption (%)
27 Jan 2004	3	3, 4	0.7–2.3	0.3–1.5	40
28 Jan 2004	1	3, 4	0.7–12.1	0.6–2.7	90
3 Feb 2004	1	7, 8	0.7–12.1	0.9–3.7	80
3 Feb 2004	2	3, 4	0.7–16.8	0.9–3.7	65
4 Feb 2004	2, 3	7, 8	1.0–13.4	1.5–3.0	80

^a Treatment number: 3, burned; 4, burned and glyphosate applied; 7, burned and grazed; 8, burned, grazed, and glyphosate applied.

visual inspection of the plots suggested that the glyphosate treatment had a significant effect on species composition. Glyphosate killed most if not all of the fountaingrass. However, the dead standing grass was still present and contributed to fuel loading 1 y following application.

The main effect of the grazing treatment did not significantly affect fuel height by itself. However, the grazing effect appeared in several significant temporal interaction terms (Table 6). The significant interaction terms with time indicate that the temporal response of fuel height to the fuel treatments differed (Figure 3b). The height of the untreated fountaingrass was relatively constant over time; the grazed plots also changed little over time. In contrast, the fuel height of the burned plots decreased significantly initially following the burns, then recovered to the pre-treatment height within 13 mo post-treatment.

Table 6. Summary of independent *F*-tests of the effect of burning, grazing, and herbicide treatments and repeated measurements on crimson fountaingrass fuel loading and fuel height near Pu'uana'hulu, Hawai'i, in 2004.

Effect ^a	Fuel loading			Fuel height		
	df ^b	<i>F</i> ^c	Pr > <i>F</i>	df	<i>F</i>	Pr > <i>F</i>
Block	2/6	1.31	0.337	2/6	2.13	0.156
Burn	1/6	26.20	0.002	1/6	20.59**	0.001
Grazing	1/6	0.00	0.972	1/6	1.84	0.196
B*G	1/6	0.07	0.804	1/6	1.49	0.242
Herbicide	1/8	3.11	0.116	1/6	10.19**	0.007
B*H	1/8	1.25	0.297	1/6	0.50	0.491
G*H	1/8	0.99	0.349	1/6	2.62	0.128
B*G*H	1/8	0.13	0.732	1/6	3.13	0.099
Time	3/48	48.15**	<0.001	3/48	37.45**	<0.001
B*T	3/48	73.93**	<0.001	3/48	76.23**	<0.001
G*T	3/48	0.51	0.674	3/48	3.71	0.018
B*G*T	3/48	1.52	0.222	3/48	4.33**	0.009
H*T	3/48	2.61	0.062	3/48	25.63	<0.001
B*H*T	3/48	1.16	0.333	3/48	2.99**	0.040
G*H*T	3/48	2.25	0.094	3/48	5.00**	0.004
B*G*H*T	3/48	2.24	0.095	3/48	2.36	0.083

^a Abbreviations: B, prescribed burning; G, cattle grazing; H, glyphosate herbicide; T, time.

^b Numerator degrees of freedom/denominator degrees of freedom.

^c *F*-statistic calculated using Type 3 (partial) sums of squares.

** *F* significant if $\alpha \leq 0.05$.

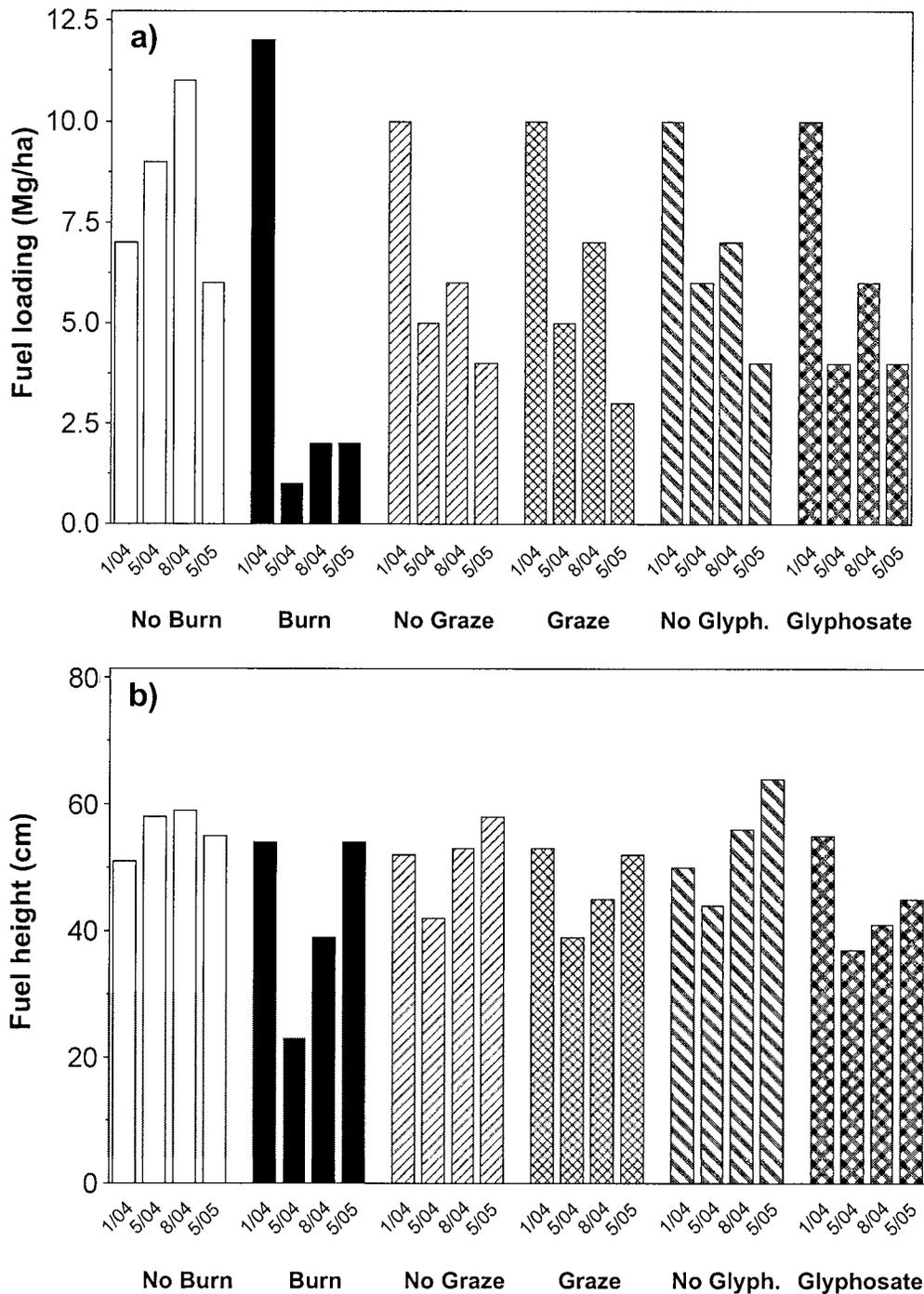


Fig. 3. Impact of prescribed burning, light cattle grazing, and glyphosate herbicide on (a) fine fuel loading and (b) fuel height to control crimson fountaingrass near Pu'u'anahulu, Hawaii, 2004–2005. The values plotted are effect means estimated as linear combinations of the least-squares estimates.

DISCUSSION AND MANAGEMENT IMPLICATIONS

The uniqueness of native Hawaiian terrestrial biodiversity is unparalleled compared to that of other oceanic island archipelagos and is recognized as one of the world's biodiversity "hotspots." Hawaii's highly varied climate, extreme geographic isolation, geological history, and broad range of habitats have interacted to form a highly diversified and unique biota. Ha-

wai's high rate of floristic endemism has provided for unique plant communities dominated by native species (Gagne and Cuddihy 1999). Tropical dry ecosystems are considered to be the most endangered ecosystem on the earth (Janzen 1988).

The glyphosate treatment resulted in good coverage and uniform kill. As a result, the glyphosate-treated split plots had well-defined edges (Figure 4). Application of the grazing treatment was not as successful as the prescribed burning or herbicide treatments.

Table 7. Summary of pairwise comparisons of estimated effect means between control and each treatment combination for fuel loading and fuel height. Significant if $\alpha \leq 0.05$. For brevity, only significant 3-factor interaction terms listed.

Treatment code ^a	Loading (Mg/ha)					Height (m)				
	Mean	Control	Reduction ^b	<i>t</i> ^c	Pr > <i>t</i>	Mean	Control	Reduction	<i>t</i>	Pr > <i>t</i>
B	4.8	8.9	46	-5.12	0.002	0.43	0.56	23	-4.54	0.000
G	6.8	6.8	0	-0.04 ns	0.972	0.48	0.52	8	-1.36 ns	0.196
Bg	4.9	8.8	44	-3.44	0.014	0.43	0.60	28	-4.07	0.001
BG	4.7	8.8	46	-3.64	0.011	0.43	0.60	28	-4.17	0.001
H	6.3	7.3	14	-1.76 ns	0.116	0.45	0.54	17	-3.19	0.007
bH	8.0	9.7	18	-2.04 ns	0.076	0.51	0.62	18	-2.76	0.015
Bh	5.0	9.7	48	-4.77	0.001	0.47	0.62	24	-3.71	0.002
BH	4.6	9.7	53	-5.16	0.000	0.40	0.62	35	-5.47	0.000
GH	6.0	7.1	15	-1.08 ns	0.300	0.41	0.54	24	-3.22	0.006
bGH	7.7	9.2	16	-1.09 ns	0.297	0.47	0.66	29	-3.24	0.006
Bgh	4.9	9.2	47	-3.08	0.010	0.42	0.66	36	-4.12	0.001
BgH	4.9	9.2	47	-3.08	0.010	0.45	0.66	32	-3.67	0.003
BGh	5.1	9.2	45	-2.97	0.012	0.51	0.66	23	-2.49	0.025
BGH	4.3	9.2	53	-3.52	0.004	0.34	0.66	48	-5.43	0.000

^a Uppercase letter denotes presence of treatment; lowercase letter denotes absence. Abbreviations: B, prescribed burning; G, cattle grazing; H, glyphosate herbicide.

^b Reduction = 100[1 - (mean/control)].

^c Student's *t*. ns, *t* is not significant if $\alpha > 0.05$.



Fig. 4. Precision of aerial spraying of glyphosate herbicide (left) to control crimson fountaingrass near Pu'uuanahulu, Hawai'i, in 2004.

Grazing intensity was light due to changing livestock ownership and ranch lease agreements at the time of treatment and difficulty in obtaining the number of animals needed to achieve the desired stocking rate. Grazing pressure was not uniform because of the high preference that the cattle displayed for the new growth in the burned plots. In addition, gates between the fenced grazing plots remained open, so grazing use and forage selection were not consistent between the different plots. It was difficult to determine the grazing pressure; therefore, the grazing results are of limited value.

Preliminary results indicated that prescribed burning is an effective tool to reduce fuel loading in fountaingrass. However, there is an element of risk associated with the use of prescribed burning. In this area of Hawai'i, there are few roads to interrupt fuel continuity. An escaped prescribed burn could easily spread 15 km to the east in <24 h under the influence of westerly onshore breezes that circulate under the predominant northeastern trade winds. At least five wildfires have burned in such a fashion over the past 20 y, threatening native plant reserves such as the Kī-pukakalawamauna Endangered Plants Area located in the Pohakuloa Training Area (Shaw et al. 1997). Because of risk of escape, the natural resources at risk, and the limited experience with prescribed fire use in this area, the per-unit cost of fire use is likely greater than in areas with less risk and more experience. The duration of the fuel load reduction caused by the prescribed burn treatment is currently unknown and cannot reliably be estimated from these data. The frequency at which prescribed fire should be applied to maintain reduced fuel loadings also influences per-unit costs.

Grazing has been shown to be an effective tool to reduce fuel loads in grass and herbaceous plants (Hutton 1920, Blackmore and Vitousek 2000). Palatability and nutritive content of the plants is an important consideration. We observed the cattle preferring the new green growth following the prescribed burns instead of the dried fountaingrass. In our study, cattle grazing was low intensity, weakly managed, and of brief duration. Cattle utilized the burned areas more than the unburned areas. Assisted by favorable growing conditions, lightly grazed areas recovered quickly. A grazing system utilizing a rotational grazing scheme has been used at Pu'uwa'awa'a, an area just west of Pu'uana'hulu, to effectively manage fountaingrass fuel loads. However, when applied as a single light pulse as in the present study, grazing does not appear to be an effective fuel reduction tool.

Glyphosate herbicide has been found to be an effective herbicide to kill fountaingrass and facilitate restoration of the dry forest on Hawai'i (Cordell et al. 2002). In our study, glyphosate applied aurally was also very effective in killing fountaingrass. However, the dead grass persisted throughout the first year and the dead bunches stayed intact. While the smaller-diameter stems had broken off, a significant proportion of standing dead fine fuels remained. Although fuel continuity has been affected, these residual dead grass

clumps remain available fuel and are more likely to support fire spread better than the living grasses with higher moisture content that they replaced. Additional time is needed to allow decomposition processes to take full effect. In order to reduce fire risk significantly, some sort of treatment following herbicide application should be considered. Continued application of the burning and grazing treatments is likely needed to maintain decreased fuel loading. Although glyphosate kills the currently growing plants, there is an adequate fountaingrass seed source to replace the dead plants. Different timings and orders of the treatments may also produce different results.

Fuel continuity, loading, and depth were altered by some of the fuel treatments. The fuel loadings were generally less than the fuel loadings reported by Wright et al. (2002) for fountaingrass in the same vicinity; Blackmore and Vitousek (2000) reported fuel loadings of 8 Mg/ha. As a result of the fuel treatments, the fuel bed bulk properties, such as bulk density, may have been altered. Blackmore and Vitousek (2000) reported increased bulk densities in grazed areas in contrast to ungrazed areas. The herbicide treatment appears to have altered the species composition and thus fuel particle characteristics such as surface area-to-volume ratio. We have begun to examine the impact of these fuel changes on potential fire behavior (Moraga 2006).

Economic analysis of the treatments used in this study is needed if these treatments are to be considered by private landowners as well as public agencies. The costs of the treatments and the time that fire risk is reduced needs to be determined in order to perform such an analysis. Unfortunately, the logistical constraints imposed on this study, such as additional fencing and extra personnel for the burns, the size of the plots used, and the economic data associated with this study, are not representative of operational costs. It is anticipated that actual treatment costs for both grazing and prescribed burning would be less than the costs in this study. However, it is difficult to extrapolate the costs to landscape levels from these small plots because of unknown economies of scale.

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USE OF PRESCRIBED FIRE AND CATTLE GRAZING TO CONTROL GUINEAGRASS

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ABSTRACT

Guineagrass (*Urochloa maxima*) is one of several species of exotic grasses that have infested rangelands in South Texas. This grass was originally introduced to improve cattle grazing productivity. However, during the past 50 y, a number of negative consequences of this introduction have occurred, including reduction of wildlife habitat quality, loss of riparian integrity, and a decrease in rangeland economic value. Exotic invasive grasses such as guineagrass can provide forage for domestic animals; however, such plants displace valuable native plants essential to wildlife and may form extensive monocultures. Prescribed burning can be used to remove old dry matter and improve forage palatability, and because cattle prefer guineagrass, it can be intensively grazed to reduce health and competitive ability. We therefore conducted summer- (June) and winter-season (February) prescribed burns in two pastures with six grazing enclosures established to compare grazing effects on burned and unburned areas. We monitored plant composition, density, cover, nonstructural carbohydrates, and nitrogen monthly for 7 mo post-burning for summer-burned and 4 mo for winter-burned sites. Guineagrass density decreased following summer prescribed burning, and native plant species richness increased. Native plant species richness in the summer burning–grazing treatment increased from 1 to 4.3 species/0.25 m², an increase of 330%. Ten important forbs used by white-tailed deer (*Odocoileus virginianus*) and six used by northern bobwhite (*Colinus virginianus*) were newly recorded or increased 1 y after application of the burning treatments. Cattle and white-tailed deer preferred burned areas.

keywords: competition, guineagrass, intensive grazing, invasion, invasive species, *Odocoileus virginianus*, prescribed burning, species richness, Texas, *Urochloa maxima*, white-tailed deer.

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INTRODUCTION

Exotic plant species are currently second only to loss of habitat as the largest threat to conservation of biodiversity (Zalba et al. 2000, Keane and Crawley 2002). Negative effects on ecological processes due to invasive plants include changes in hydrological processes, erosion potential, sedimentation, energy flow, nutrient cycling, regeneration of native plants, and fire regimes (Masters and Sheley 2001). Guineagrass is one of several species of exotic grasses that have infested rangelands in South Texas. These grasses were originally introduced to improve cattle grazing productivity. Exotic invasive grasses provide forage for domestic animals; however, such plants displace valuable native plants essential to wildlife and may form extensive monocultures. Negative consequences of invasive plants include reduction of wildlife habitat quality, loss of riparian integrity, and a decrease in rangeland economic value (Masters and Sheley 2001).

Fire has increasingly gained acceptance as a management tool (Wright and Bailey 1982). In the case of invasive species such as guineagrass and other forages, one of the most important effects of prescribed fire is the removal of old growth and stimulation of regrowth. Guineagrass regrowth is thus very palatable for cattle compared to native plant species regrowth; therefore, grazing pressure would be higher on guineagrass, favoring the growth and reestablishment of native spe-

cies in an area invaded by guineagrass. In addition to the effect of fire on cattle preference for guineagrass, white-tailed deer (*Odocoileus virginianus*) also prefer sprouts of many brush species (Fulbright and Ortega 2006) and forbs that occur in burned areas; therefore, concentration of white-tailed deer is expected in burned areas. Intensive grazing is a well-known strategy for managing invasive introduced grasses; however, most of the published information discusses combinations of factors to balance defoliation intensity and plant reserves to achieve persistence and maximum forage productivity (Marachin et al. 1981, Jones 1986, Canudas-Lara 1988). Grazing strategies can be used to deplete carbohydrate reserves, reduce tillering and meristem tissue development, and kill undesirable invasive plants.

Our study objective was to evaluate the effect of the combination of intensive cattle grazing and prescribed burning on the survival of guineagrass and to quantify the effect of these practices on reestablishment of the native plant community and preference of white-tailed deer for burned areas.

MATERIALS AND METHODS

Study Site

We conducted this project in the Rio Grande Sand Plains (Blair 1950) of South Texas in Willacy County, where two 100- and 200-ha pastures were used at Te-colote Ranch. The climate was subtropical humid with

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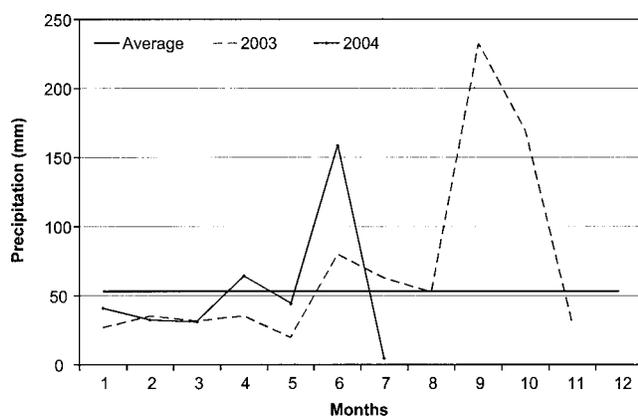


Fig. 1. Monthly rainfall at Tecolote Ranch, near Raymondville, Texas, during 2003 and from January to July 2004.

hot, dry summers and mild winters, with a frost-free season of >300 d. Mean monthly temperatures ranged from 15°C in the winter to 29°C in the summer, but summer temperatures exceeding 35°C were typical. Precipitation was erratic, with 6–8 mo of the year relatively dry, and mean annual rainfall of 650 mm. Rainfall during 2003 and 2004 was unusually high compared to the long-term average. During the months of September and October 2003, rainfall was 233 and 169 mm compared with the long-term average of 137 and 80 mm, respectively (Ramirez 2005) (Figure 1). Scifres and Hamilton (1993) classified the vegetation as a scrubland complex with summer green arborescent, green thorn scrub, and dry–mesic summer green components. The mesquite (*Prosopis glandulosa*)–granjeno (*Celtis ehrenbergiana*) association was the major scrubland association. Study site soils were predominantly sandy (Delfina loamy fine sand and Nueces fine sand) and varied from alkaline to slightly acid (Pendleton and Carter 1974, Nelle 1982). Two guineagrass-dominated pastures mixed with native vegetation with similar guineagrass density were selected for the burning and grazing treatments. Prior to the study, pastures were grazed with a low cattle stocking rate (1 animal unit/14 ha).

Treatments

Eight treatments with three replications resulting from the combination of two seasons (summer and winter) and two factors were evaluated: prescribed burning (burning and no burning) and intensive grazing (grazing or no grazing). The following treatments were applied in summer (June) and winter (February): burning–grazing, burning–no grazing, no burning–grazing, and no burning–no grazing. The experimental site was closed to grazing to accumulate at least 2,000 kg/ha of fine fuel before the application of the burning treatments. Prior to burn application, a double sampling technique (Frame 1981) was used to estimate fine fuel accumulation to ensure at least 2,000 kg/ha was available. Prescribed burning was applied in June 2003 (summer) and February 2004 (winter). Electric fencing was used to protect the no-grazing treatments. Each plot of the no-grazing treatments was about 0.5

ha. After burning, pastures including intensive grazing were grazed whenever guineagrass reached 30 cm in height and at the end of the growing season when the grass began flowering. Cattle were removed from pastures after consumption of 75% of guineagrass standing biomass, which was determined by sampling biomass before grazing using a double sampling technique (Frame 1981) for the pasture and estimating the time cattle would need to consume 75% of the standing biomass.

Field Measurements and Laboratory Analysis

Pre-treatment data to determine density, basal cover, and foliar cover of guineagrass and native plant species foliar cover and species richness were collected, using a 25 × 50-cm Daubenmire frame (Chambers and Brown 1983). We monitored guineagrass density, foliar cover, plant species richness, total nonstructural carbohydrates, and nitrogen in stem bases monthly for 7 mo post-burning for summer burns and 4 mo for winter-burned sites to identify plant response to treatments. Two plots per treatment were used and 15 sampling points were established in each plot.

Total nonstructural carbohydrate concentration levels in stem bases was measured monthly to monitor plant reserves by acid hydrolysis as described by Murphy (1958) and by Kjeldahl procedure (Kjeldahl 1883) for nitrogen from December 2003 to June 2004. Samples were collected and immediately placed on ice and transported to the Lehmann Research Laboratory of the Animal and Wildlife Sciences Department at Texas A&M University-Kingsville in Kingsville, Texas, for chemical analysis. Upon return to the laboratory, plant samples were dried in a forced-air oven at 60°C for 3 d. Subsequent to drying, soil was removed from samples and the roots were dissected from basal crown and stem bases. The stem bases, including the lower 2–2.5 cm of stem, and basal crown were dissected from the rest of the plant shoot and root material, ground in a Willey mill to pass through a 0.5-mm screen, and stored in amber vials until analysis.

A cattle survey was conducted to measure cattle preference between burned and non-burned areas. Number of animals observed in the burned and non-burned areas was recorded 2 times/d (at 0800 and 1800) 2 d every month. To evaluate white-tailed deer preference for burned and non-burned areas, deer spotlighting surveys were conducted on the winter-burned and non-burned areas every month. The same 3-mi (4,827-m) transects were used each month to record deer presence in the burned and non-burned plots. The summer-burned area was not surveyed because the size and distribution of burned areas were not sufficient to separate differences between burned and unburned areas.

Statistical Analysis

Data were analyzed using a randomized complete block design with a factorial arrangement of treatments. Main effects and interactions were considered significant at $\alpha < 0.05$. When significant main effects

Table 1. Guineagrass mean foliar cover under different combinations of prescribed burning and intensive grazing at Tecolote Ranch, Raymondville, Texas, 2003–2004.

Treatment ^a	Season			
	Summer (Jun)		Winter (Feb)	
	% cover	SE	% cover	SE
BG	53	1	79	6
BNG	18	1	98	1
NBG	63	9	58	11
NBNG	56	11	85	8

^a Treatment: BG, burning–grazing; BNG, burning–no grazing; NBG, no burning–grazing; NBNG, no burning–no grazing.

or interaction effects were found, we assessed difference among treatment means using Duncan's test procedure at $\alpha = 0.05$. Cattle preferences during the trial were analyzed by comparing burned and non-burned areas with analysis of variance. White-tailed deer habitat use was compared by regression analysis. All statistical analyses were done with STATISTICA[®] software (StatSoft 2004).

RESULTS

Pre-treatment Data

Pre-treatment average vegetation data were analyzed for guineagrass density (80,000 plants/ha), guineagrass basal cover (12%), guineagrass foliar cover (70%), and native plant species foliar cover (15%) and species richness (1.1 species/0.25 m²). We detected no significant differences ($P > 0.05$) among treatments for any of these parameters.

Guineagrass Density

Prescribed burning reduced ($F = 6.863$, $P = 0.058$) guineagrass density in summer-burned areas, with an average of 58,667 plants/ha (SE = 11,366, $n = 2$) compared to 93,333 plants/ha (SE = 10,442, $n = 2$) in non-burned areas. Grazing did not affect ($F = 0.379$, $P = 0.542$) guineagrass density in summer- or winter-burned areas. Prescribed burning increased ($F = 4.452$, $P = 0.043$) guineagrass density in the winter-burned areas, with an average of 115,556 plants/ha (SE = 7,499, $n = 9$) in burned areas compared to 94,222 plants/ha (SE = 11,769, $n = 9$) in non-burned areas.

Guineagrass Foliar Cover

In summer-burned areas, mean guineagrass foliar cover was 18% (SE = 1.7%, $n = 2$; $P = 0.005$) in the burning–no grazing treatment compared to 53% in the burning–grazing (SE = 1.2%, $n = 2$), 63% in the no burning–grazing (SE = 8.9%, $n = 2$), and 56% in the no burning–no grazing (SE = 11.1%, $n = 2$) treatments (Table 1). Guineagrass foliar cover decreased from 38% to 17% in the burning–no grazing treatment by the end of study ($P > 0.05$). In winter-burned areas, guineagrass foliar cover was affected by both burning ($F = 4.679$, $P = 0.038$) and grazing ($F = 6.289$, $P = 0.017$). Guineagrass foliar cover was lower in non-

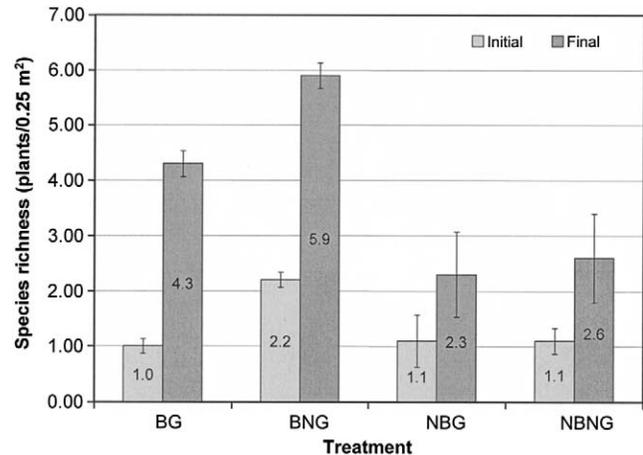


Fig. 2. Final native plant species richness (plants/0.25 m²) 12 mo after summer prescribed burning and subsequent intensive grazing in a guineagrass-dominated pasture at Tecolote Ranch near Raymondville, Texas, 2003–2004. Treatments: BG, burning–grazing; BNG, burning–no grazing; NBG, no burning–grazing; NBNG, no burning–no grazing.

burned areas, with an average of 71% (SE = 10.3%, $n = 9$) compared to 89% (SE = 5.2%, $n = 9$) for prescribed burn treatments. Guineagrass foliar cover was lower in grazing treatment areas, with an average of 68% (SE = 9%, $n = 9$) compared to 92% (SE = 6.3%, $n = 9$) for no-grazing treatments.

Native Plant Species Richness

Sixty-four plant species occurred in the sampling plots where at least one treatment (burning or grazing) was applied. Higher native plant species richness was observed in summer-burned areas compared to winter-burned. Plant species richness in the summer burning–grazing treatment increased 330% from 1.0 to 4.3 species/0.25 m², increased 150% from 2.2 to 5.9 species/0.25 m² in the burning–no grazing treatment, increased 109% from 1.1 to 2.3 species/0.25 m² in the no burning–grazing treatment, and increased 136% from 1.1 to 2.6 species/0.25 m² in the no burning–no grazing treatment (Figure 2). Plant species richness differed ($F = 22.086$, $P = 0.009$) among burning treatments and was lower in no-burning areas, with an average of 2.5 (SE = 0.5, $n = 2$) compared to 5.2 species/0.25 m² (SE = 0.5, $n = 2$) for burning treatments. Species richness was not affected ($F = 2.7$, $P = 0.175$) by grazing. When data were analyzed to determine the percentage change of species richness between the beginning and end of the study, the interaction between prescribed burning and grazing was significant ($F = 9.56$, $P = 0.036$). Ten forbs used as food by white-tailed deer were newly recorded or increased cover after 1 y in the burning treatments (Table 2). Six important food plants for northern bobwhite (*Colinus virginianus*) were observed in burned treatments (Table 2) but were not found in unburned areas.

In the winter-burned areas by the end of the study, plant species richness ranged from 0 to 2.3 ± 0.84 species/0.25 m². Species richness was different ($F =$

Table 2. Native plants used by white-tailed deer and northern bobwhite in burned areas of guineagrass-dominated pastures 1 y after prescribed burning, Tecolote Ranch near Raymondville, Texas, 2004.

Animal species	Plant species	
	Scientific name	Common name
White-tailed deer	<i>Aphanostephus skirrhobasis</i>	Lazy daisy
	<i>Callirhoe digitata</i>	Winecup
	<i>Commelina erecta</i>	Dayflower
	<i>Desmanthus virgatus</i>	Desmanthus
	<i>Lesquerella argyraea</i>	Silver bladderpod
	<i>Oxalis dillenii</i>	Woodsorrel
	<i>Chamaesyce glyptosperma</i>	Ridgeseed spurge
	<i>Plantago rhodosperma</i>	Redseed plantain
	<i>Rhynchosia americana</i>	American snoutbean
Northern bobwhite	<i>Verbena officinalis</i>	Verbena
	<i>Chamaecrista fasciculata</i>	Partridge pea
	<i>Commelina erecta</i>	Dayflower
	<i>Coreopsis nuecensoides</i>	Coreopsis
	<i>Digitaria arenicola</i>	Witchgrass
	<i>Setaria parviflora</i>	Bristlegrass
	<i>Sporobolus indicus</i>	Smutgrass

9.603, $P = 0.004$) among burning treatments. Richness was lower in burning treatments, with an average of 0.1 species/0.25 m² (SE = 0.08 species/0.25 m², $n = 9$) compared to 1.8 species/0.25 m² (SE = 0.75 species/0.25 m², $n = 9$) for no-burning treatments. Species richness was not affected by grazing ($F = 1.685$, $P = 0.204$).

Total Nonstructural Carbohydrates and Nitrogen

No significant ($P > 0.05$) differences were found among treatments for total nonstructural carbohydrates or nitrogen in the summer- or winter-burned and no-burned areas. No difference ($P > 0.01$) was found for total nonstructural carbohydrates or nitrogen between the summer- or winter-burned areas.

Cattle and White-tailed Deer Preference

Initially (February 2004), white-tailed deer presence was higher ($P = 0.033$) in the no-burning area, with 19 animals versus 5 animals/3-mi (4,827-m) transect in the burned area. During the next 4 mo, deer presence in no-burning areas gradually decreased ($y = 14.5 - 1.63x$, where $x =$ time in months after burning) and finally just 2 deer were observed. The opposite ($P = 0.008$) behavior was observed in the burned areas, where the presence of deer increased to 14 deer ($y = 4.93 + 1.07x$, where $x =$ time in months after burning; $R^2 = 63.9$). Cattle highly preferred burned areas ($P = 0.001$), with an average of 44.5 animals (SE = 2.7 animals, $n = 26$) compared to an average of 32.5 animals (SE = 2.4 animals, $n = 26$) in non-burned areas.

DISCUSSION

In general, summer burns had a more dramatic effect on guineagrass in terms of density, cover, and native plant species richness compared to winter burns. Summer prescribed burning reduced guineagrass density independently of the grazing treatment. These results agree with those of Skovlin (1971), who indicated that guineagrass is susceptible to hot fires. However, in that study, no change was observed when post-treatment guineagrass density was compared to pre-treatment density. In our study, grazing did not significantly affect guineagrass density, in contrast to several studies that indicate high grazing pressure, in general, reduces grass density (Watkin and Clements 1978, Santillan 1983, Valentine 1990).

Native plant species richness, an index to reestablishment of the native plant community, was higher on summer-burned areas compared with areas burned in winter. Species richness in the burning–no grazing treatment increased 150% from 2.2 to 5.9 species/0.25 m², and in the burning–grazing areas increased 330% from 1.0 to 4.3 species/0.25 m². This result contrasts with that of Drawe and Kattner (1978), who indicated that percent composition of grasses and forbs was not significantly affected by early summer prescribed burning when combined with mowing. In our study, summer burns appeared to improve species richness, perhaps because of a significant reduction of guineagrass foliar cover. In a similar fashion, Santillan (1983) found a short rest period improved legume yields and development after heavy grazing when associated with guineagrass and elephant grass (*Pennisetum purpureum*)—this was due to reduction of grass competition for light. However, Davison and Brown (1985) reported that “Gatton” panicgrass (guineagrass, i.e., *Panicum maximum*) can compete versus legumes after 4 y of heavy grazing pressure, due to legume yield reduction after 4 y. Others have reported that after 3 y of grazing, species richness improved compared with control areas (California Agricultural Technology Institute 2004).

Comparison between grazing and no-grazing plots in winter-burned areas showed a 22% decrease on guineagrass foliar cover, and that extra space was occupied by a 7% increase in native species cover and a 15% increase in bare ground. Different results were reported by Duval and Linnartz (1967), who stated that bare ground did not change 1 y after burning and grazing, but an increase in total herbage yield and species richness was recorded in following years.

White-tailed deer and cattle habitat use was influenced by prescribed burning. A higher concentration of cattle in the burned areas was recorded throughout the study. New regrowth and nutritious forage on burned and frequently grazed areas maintained higher cattle densities. This result agrees with those of Ralphs et al. (1995), Howery et al. (1998), and Villalba and Provenza (1999), who reported livestock preference was highly influenced by forage quality, which is usually higher on grass regrowth. White-tailed deer use decreased immediately after burning, probably because

of decreased food sources (Davis 1990). Forbs and brush are the main components of deer diet and are highly affected by fire; however, recovery is slower than that of grasses (Payne and Bryant 1998). One month after burning, deer numbers gradually increased in burned areas and decreased in unburned areas. The increase in native plant species richness and the nutritious regrowth of brush species in the burned areas positively affected the preference of white-tailed deer for burned sites.

Our study results were possibly affected by two factors: 1) rainfall pattern during the period of study, and 2) time needed to achieve grazing effect on the plant community. Two months after summer prescribed burning, heavy rain fell on the study area (Figure 1), 233 mm during September and 169 mm during October 2003. The average precipitation for these months is 137 mm and 80 mm for September and October, respectively, which was 70% and 111% higher than the average, respectively (Ramirez 2005). This unusually high rainfall may have allowed guineagrass to grow and recover from the grazing treatment, increasing mineralization and deposition of ash and atmospheric nitrogen (USGS 1999, Sadras and Baldock 2003). Sadras and Baldock (2003) reported that frequent rainfall events can mineralize >70 kg N/ha when seasonal precipitation exceeds 300 mm, which we observed in this study. USGS (1999) also reported that 30 mm of rainfall can deposit 168 g N/ha directly from the atmosphere in the South Texas Coastal Bend area. This suggests that, in this study, guineagrass could have received >70 kg N/ha. Increases in biomass production of guineagrass have been reported by the Food and Agriculture Organization (2003) as a result of nitrogen fertilization. Middleton and McCosker (1975) reported biomass production of guineagrass of 60 t of dry matter/ha when 300 kg of N was applied in North Queensland, Australia. Secondly, plant longevity can be reduced by heavy grazing but usually after a period of time, the grazing effect over grass persistence occurs after 3 y of intensive grazing (Jones 1986). Jones (1986) also reported African brittlegrass (*Setaria sphacelata*) density dropped from 59,000 to 17,000 plants/ha after 2 y and to 9,000 plants/ha after 8 y. Some other studies report reduction of panicgrasses only after long periods (more than 3 y) under high grazing pressure (Duval and Linnartz 1967). In our study, intensive grazing was applied for only 1 y and this time period appears to be insufficient to reduce guineagrass density.

MANAGEMENT IMPLICATIONS

The application of prescribed burning and subsequent intensive grazing during the summer altered plant species composition on pastures dominated by guineagrass. Guineagrass density decreased with summer prescribed fire. The percent change in plant species richness by the end of the study was higher in the burning–grazing treatment during the summer; but a longer period of time is required to assess the desired

effect of grazing and burning on guineagrass density. This initial study suggests burning and grazing can lead to an increase in native plant species, even when extremely wet years apparently masked the effect of grazing on guineagrass density and high precipitation positively affected guineagrass survival. Cattle preferred burned areas probably because of increased guineagrass forage quality. The increase in native plant species richness and the nutritious browse available after burning may have caused the high preference of white-tailed deer for burned areas.

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THE RESPONSE OF OLD WORLD BLUESTEM TO MID- AND LATE-SUMMER FIRE IN TWO GRASSLAND ECOREGIONS IN CENTRAL TEXAS

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ABSTRACT

Old World bluestem (*Bothriochloa ischaemum*) is an invasive, nonnative, C₄ perennial grass that has become widespread throughout the southern United States. It is considered a problematic invader of fields and roadsides in north-central, central, and southern Texas. Although the growth of this species is enhanced by dormant-season prescribed fire, observations suggested that this species might prove sensitive to growing-season prescribed fire.

Identical suites of treatments were installed at two locations in central Texas representative of two grassland ecoregions: Edwards Plateau and Blackland Prairie. Frequency measurements taken during the growing season in the year following treatment demonstrated that *B. ischaemum* was significantly reduced at both Edwards Plateau ($F = 26.99$, $P < 0.0001$) and Blackland Prairie ($F = 27.11$, $P < 0.0001$) sites. The magnitude of the response was not consistent, however, with a reduced response at one site where there had been a significant rainfall immediately before and after the burn installation. We suggest that successful selective control of this species may be achieved by synchronizing growing-season prescribed fire during dry conditions.

keywords: *Bothriochloa ischaemum*, grassland, invasive, Old World bluestem, prairie, prescribed fire, Texas.

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INTRODUCTION

Old World (yellow or King Ranch) bluestem (*Bothriochloa ischaemum* var. *songarica*) is a C₄ (warm-season) perennial bunchgrass native to Asia and Central Europe that has become widespread throughout the southern United States since its introduction in the early 1900s (Gould 1978). Although the benefits of Old World bluestem as a cattle forage crop have been recently called into question (Coleman et al. 2001), it has been successfully used for livestock range improvement throughout the southern Great

Plains (Coyne and Bradford 1984, Berg et al. 1996, Teague et al. 1996), including marginal rangeland in Oklahoma and Texas (Berg 1993, Welch et al. 2001). Although the precise distribution has not been quantified, Old World bluestem is presently considered a problematic invader of temperate grasslands in North America (Grace et al. 2001, Harmoney et al. 2004) and of roadsides in north-central (Diggs et al. 1999), central, and southern Texas (Union of Concerned Scientists 2003). Typical of other invasive exotic species, Old World bluestem has the ability to crowd out native plants (Diggs et al. 1999), threaten rare native plant species (Union of Concerned Scientists 2003), and suppress insect and small-mammal diversity (Schwertner 1996, McIntyre and Thompson 2003).

The spread of Old World bluestem into native

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grasslands has been the result of the combination of widespread agricultural and roadside seeding (Gould 1975), negative grazing selection by livestock, its relative ease of establishment, and ability to successfully compete with native grasses over the long term (Eck and Sims 1984, Berg 1993). Old World bluestem is a highly stress-tolerant C_4 grass, with high carbon balance and water use efficiency under marginal abiotic conditions of low moisture and nitrogen (Szenete et al. 1996). Under grazing pressure, this species allocates more resources to basal structures and switches from an upright to more prostrate form (Campanella 1977). However, it does exhibit sensitivity to interspecific competition under continual grazing or mowing (Szenete et al. 1996) and drought sensitivity in the seedling stage (Berg et al. 1996). Additionally, there have been indications that while dormant-season (winter) fire may increase production of Old World bluestem in the subsequent growing season (Pase 1971), early growing-season (spring) fire can have a negative effect on growth (Berg 1993).

Studies examining the responses of plant populations to seasonality of fire in grasslands are sparse and somewhat inconclusive (Towne and Owensby 1984, Glenn-Lewin et al. 1990, Howe 1994b, Engle and Bidwell 2001). However, what emerges is that although many species seem unaffected by fire seasonality, those that are sensitive demonstrate species-specific rather than guild-level (C_3 grass, C_4 grass, summer forb, etc.) responses (Wright and Klemmedson 1965, Mayeux and Hamilton 1988, Howe 1994b, Kush et al. 1998, Sparks et al. 1998). These differential responses suggest that utilization of seasonally applied prescribed fire need not negatively impact the growth of all species within the plant community, or even all species within a single guild. Therefore, seasonally applied prescribed fire could be used as a selective filter for the removal of unwanted sensitive species.

The objective of this study was to evaluate the effectiveness of mid- and late-growing season fire for suppressing the growth of Old World bluestem on sites representative of two Texas grassland ecoregions: Blackland Prairie and Edwards Plateau.

STUDY AREA

Two sites in Texas were selected for this treatment: one representative of Blackland Prairie and the other representative of the Edwards Plateau ecoregion. The Blackland Prairie site was located at the San Marcos National Fish Hatchery and Technology Center in San Marcos, Texas (29°50'15"N, 97°58'45"W; elevation 300 m). Soils were fine, montmorillonitic, thermic Udic Pellusterts of the Houston Black series, characteristic of upland Blackland Prairie in central Texas (USDA 1984). The site was donated to the U.S. Fish and Wildlife Service in the 1960s; however, its prior management history is unknown. Prior to treatment, the pasture was largely dominated by Old World bluestem (39% cover) and silver bluestem (*B. laguroides*) (19%), with a mixture of native and nonnative forbs.

The Edwards Plateau study site was at the Lyndon B. Johnson National Historical Park, Johnson City, Texas (30°16'16"N, 98°25'02"W; elevation 375 m). Soils were very fine, montmorillonitic, thermic Udic Chromusterts of the Anhalt series, which are limestone-derived clayey soils typical of flat upland sites on the Edwards Plateau (USDA 1979). The area had been maintained as rotated cattle pasture for the last decade. Dominant grasses included Old World bluestem (55%), Texas wintergrass (*Nassella leucotricha*) (16%), and meadow dropseed (*Sporobolus compositus*) (16%). Taxonomic nomenclature follows Kartesz (1999).

The climate for both sites was subtropical, sub-humid: mild winters and hot summers with a bimodal (May and September) rainfall pattern. The 30-y, monthly mean temperature range and average annual rainfall were 16.2–35.2°C and 878 mm, respectively, for the Blackland Prairie site, and 4.5–35.6°C and 844 mm, respectively, for the Edwards Plateau site. Climate data were obtained through National Weather Service database and from observation from nearby weather stations.

METHODS

Five replicates of three treatments (mid- and late growing-season fire and no fire) were installed at both sites following a randomized plot design. Each experimental unit measured 15 × 15 m, separated by a 10-m mowed boundary used for accessibility and to serve as a firebreak. All experimental units were ignited using ring-firing technique following wet lining along the firebreaks. Weather parameters were taken on-site using belt weather kits.

Species canopy frequency of all species was assessed using a single diagonal, corner-to-corner transect for each experimental unit, with point-intercept readings taken every 1 m (20 points per unit). Only living vegetative or reproductive tillers were recorded. Frequency of Old World bluestem and other dominant species was measured prior to treatment (May 2004) and again the following year (May 2005). The effect of treatment on frequency of Old World bluestem was examined with analysis of covariance, with pre-treatment species frequency as the covariable. Although other species within the plant communities at both sites had low or patchy distributions, Texas wintergrass and meadow dropseed had adequate occurrences (i.e., with few null data points) to also allow parametric analysis. Tukey multiple comparisons were used to test for differences between means. An alpha of 0.05 was used throughout.

RESULTS

In central Texas the growing season for Old World bluestem and for most C_4 grasses extends from mid-May through to mid-November in a normal year (Gould 1975). The timing of mid- and late growing-season prescribed fire was dictated by weather and lo-

Table 1. Weather, rainfall patterns, and fuel conditions for mid- and late growing-season prescribed fires conducted at Blackland Prairie and Edwards Plateau sites, central Texas, during 2004.

Site	Season	Date	Mean air temperature (°C)	Eye-level wind speed (km h ⁻¹)	Mean relative humidity (%)	Cumulative rainfall (mm)						Fine fuel load range (kg ha ⁻¹)
						No. days pre-burn			No. days post-burn			
						5	10	20	5	10	20	
Blackland Prairie	Mid-season	13 Aug	32	8–16	80	1	2	28	0	8	40	1,900–2,400
	Late-season	8 Nov	26	5–11	60	0	36	121	0	110	112	1,900–2,400
Edwards Plateau	Mid-season	25 Aug	31	8–16	65	8	21	47	3	48	51	1,200–1,900
	Late-season	28 Oct	28	10–13	50	61	65	94	69	71	71	1,200–1,900

gistical constraints, such that the burn events in a given season were not coincident for Blackland Prairie and Edwards Plateau sites (Table 1). However the conditions for each prescribed fire were somewhat similar in terms of temperature, wind, and humidity (Table 1), with low-intensity fire fronts with flame heights <1 m but >90% total canopy burn. Fine fuel loads (visually estimated) were different both among experimental units and particularly between sites. Lower fuel loads at the Edwards Plateau site were considered to be a result of only 6 months' rest from moderate grazing pressure (Table 1). Although annual rainfall totals for both sites through the experimental period (4–5 May) were similar (Blackland Prairie: 1,299 mm; Edwards Plateau: 1,290 mm), rainfall totals during the days immediately prior to and after fire installation were variable (Table 1). The late-season prescribed fire at the

Edwards Plateau site experienced relatively high cumulative rainfall during the 5 d before (61 mm) and after (69 mm) the day of the burn (Table 1). All other sites experienced <10 mm for the same time intervals before and after the burn (Table 1). Three of the four burn treatments reduced cover of Old World bluestem from 60–70% to <10% canopy cover. However, the late growing-season treatment at the Edwards Plateau site exhibited a reduced response of only 34% (Figure 1). Analysis of the other two subdominant species, Texas wintergrass and meadow dropseed, indicated neither a positive or negative response to treatment (data not shown).

DISCUSSION

Prescribed fire applied during the growing season resulted in reduced canopy cover of Old World bluestem in the following year (Figure 1). However, the magnitude of the response was not consistent, with the late-season fire at the Edwards Plateau site having a reduced effect. Elsewhere, variation in plant mortality has been attributed to the coincidence of fire characteristics and plant resource allocation patterns (Wright and Klemmedson 1965, McDaniel et al. 1997). We suggest that the reduced impact of the fire on one out of four treatments was a consequence of significant rain events immediately before and after the fire event. Fire intensity may have been reduced by higher dead fuel moisture, enhanced water status of the target species may have reduced tissue damage during the fire, or damaged plant individuals may have been more likely to recover due to high soil moisture after burning (Table 1; Figure 2).

Fire can directly affect individual growth and regeneration conditions negatively or positively (Baruch and Bilbao 1999) by influencing post-disturbance conditions and competitive species interactions (Hartnett 1991, Bond and van Wilgen 1996, Suding 2001). The direct effects of fire, such as tissue and whole-plant death, are complex, with many biotic and abiotic variables (Bond and van Wilgen 1996). In general, theory suggests that an individual plant is more likely to benefit from the various fire effects if the timing is asynchronous with the plant's active stages of growth when the risk of living tissue damage is reduced (Howe 1994b, Copeland et al. 2002). The converse argument—that growing-season fire will negatively affect plant growth—is not necessarily true. Species sensitivity to fire during the growth phase is species-specific

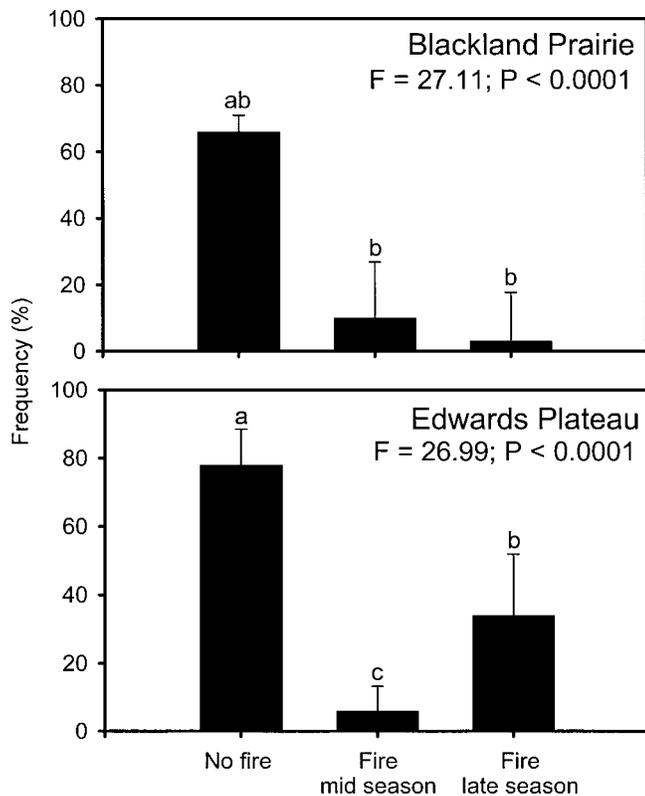


Fig. 1. Canopy cover response of Old World bluestem to mid- and late growing-season fire at Blackland Prairie (top) and Edwards Plateau (bottom) sites, central Texas, 2004. Error bars represent 95% confidence limits. Bars with different letters have means that are significantly different at $P < 0.05$.

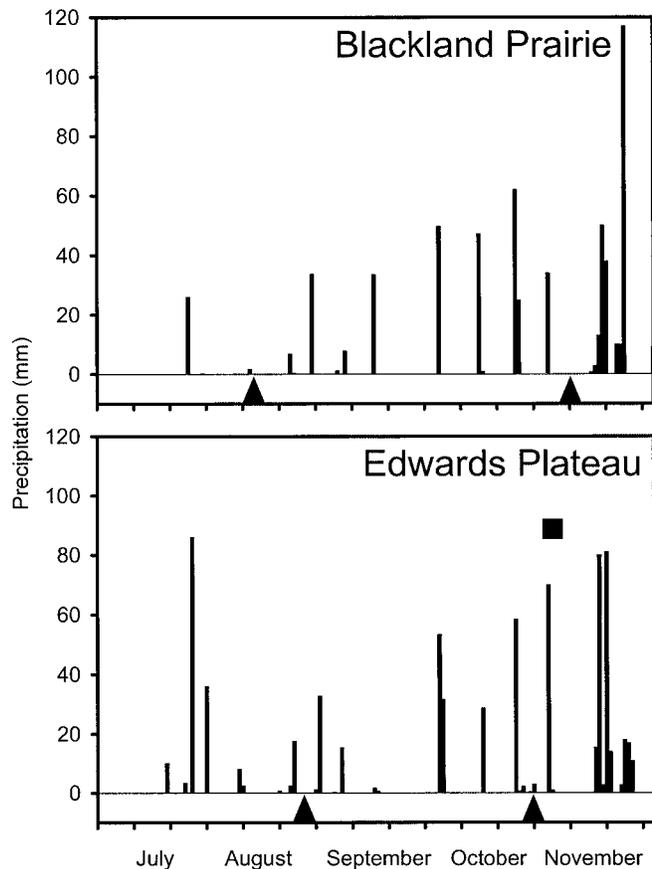


Fig. 2. Daily precipitation during the 2004 growing season at Blackland Prairie (top) and Edwards Plateau (bottom) sites, central Texas. Solid triangles indicate the days of prescribed burns.

and may be negative, neutral, or positive (Wright and Klemmedson 1965, Frost and Robertson 1985, Howe 1994a, Whelan 1995, Bond and van Wilgen 1996). Although some studies have indicated that grass mortality is reduced in more mesic sites (Zedler and Loucks 1969) and during wetter years (Frost and Robertson 1985), there seems no direct evidence for the mechanism explaining the variation of tissue damage due to available water. An understanding of the mechanism would help application of this technique and aid prediction of grass responses for years with different precipitation patterns.

Regardless of mechanism, and given the limited time period of the experiment, the target species in this study appears to have been prone to fire damage during the mid- and late growing season, which is in sharp contrast to the effect of dormant-season fire (Pase 1971). The neutral response from Texas wintergrass and meadow dropseed may support the hypothesis that many grassland species are fire adapted. Alternatively, this phenomenon may be a simply a artifact of low occurrence of these species. Given that many other dominant native perennial grass species in Great Plains grasslands have demonstrated positive or indifferent response following growing-season fire (Towne and Owensby 1984, Sparks et al. 1998), this study indicates growing-season fire may be used to selectively

reduce undesirable invasive species without eliciting a detrimental response in desirable native species.

MANAGEMENT IMPLICATIONS

Theory concerning the responses of grassland species to fire has been largely based on the results of dormant-season prescribed fire. Consequently, it is not uncommon for land practitioners to conclude that fire responses in grasses are consistent regardless of season. We suggest that such generalizations of plant guild response to fire may be inaccurate and species-specific responses to fire season require further examination. Understanding these fires' response characteristics may have particular significance for control of invasive species in grasslands.

Species response to fire may depend on timing with respect to climate and phenological stage of the individual plant. Although Old World bluestem is suppressed by growing-season fire, many native grasses are not. Therefore, timely application of growing-season prescribed fire offers an alternative to other control methods, which can help land managers selectively control the spread of this invasive species while sustaining more desirable native grasses.

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THE REHABILITATION OF NATIVE VEGETATION COMMUNITIES IN AN URBAN SHRUBLAND SETTING: THE ROLE OF CONTROLLED FIRE REGIMES

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ABSTRACT

Throughout Australia's biogeographical history, the vegetation over most of the continent has experienced a long and complex evolutionary relationship with fire. Many of the vegetative forms that have evolved over time, and which are uniquely Australian, rely on fire to complete one or more stages of their life cycle.

In recent years, changes to the natural fire regimes and the introduction of many new plants throughout Australia have led to changes in ecosystem functioning and species composition in many Australian shrubland areas. Research was undertaken to determine if regular low-intensity burning of urban shrubland areas can assist in the rehabilitation of native species by imposing a fire regime that will increase their abundance and diversity and decrease the abundance and diversity of invasive weed species. To test this hypothesis, sampling was undertaken over an 18-mo period in eight sites with four differing fire histories—3, 6, 8, and 10 y since the last fire. Within each of these sites, species presence and abundance were analyzed along two 100-m transects. Seed bank samples were also taken at these sites. Seeds within these samples were exposed to temperatures and smoke products present in a low-intensity burn and then germinated to determine the species composition of the next generation. Nutrient levels were measured to determine if any changes in species composition could be explained by elevated nutrient levels rather than by a specific fire regime.

The results from this study indicate that there is a peak in native abundance and diversity with an associated decrease in exotic abundance and diversity approximately 6 y after a low-intensity burn. There was no significant difference found in nutrient levels between sites, indicating that elevated nutrient levels did not play a role in the differences in vegetation composition. Seed germination showed significant variation in sites with different fire histories, demonstrating that time since last fire plays an important role in the species composition of the next generation of vegetation post-fire. This conclusion has many applications in the management of urban shrubland areas throughout the Sydney region. Determining the optimal frequency and intensity of fires in these areas provides much needed information for determining the best possible burning regimes for the ecology of native species.

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RELATIONSHIPS BETWEEN FIRE, SOIL WATER, AND INVASION OF SEMIARID SHRUBLAND BY AN EXOTIC FORB

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ABSTRACT

Persistent infestation by exotic forbs is common following fire in sagebrush steppe of southeastern Idaho, and in other cold desert shrublands of the Great Basin, USA. Whereas disturbances such as fire are commonly cited causes of exotic plant invasions, few studies clarify how fire alters site conditions to favor long-term persistence of exotic forbs, such as leafy spurge (*Euphorbia esula*). We hypothesized that fire increases deep soil water by selectively removing deep-rooted woody plants and that increases in the availability of deep soil water are key to the success of leafy spurge. We measured soil water contents, and depth of water uptake, water status, and photosynthetic water use of leafy spurge and native plants in a replicated set of burns. Soil water increased substantially below about 0.4-m depth in burned compared to control plots, due largely to removal of deep-rooted sagebrush. Spurge had no greater photosynthesis or water status than native plants. However, leafy spurge acquired water from deeper depths than native herbs, as revealed by comparisons of isotopic compositions of water in plants and soil at different depths. Sagebrush also acquired water from deeper depths than native herbs, and leafy spurge rarely occurred in unburned plots that had sagebrush. These data suggest that leafy spurge might experience less carbon uptake and water status than native herbs without access to the fire-induced increases in water availability in soils beneath rooting zones of native herbs. Many other exotic forbs in semiarid shrub and grasslands of western North America have deep roots and other traits in common with leafy spurge, and site resistance to dominance by exotic forbs may be enhanced by fire prescriptions that favor shrub reestablishment (e.g., patch burns) or restoration efforts that promote deep-rooted native species.

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FIRE LEARNING NETWORK IN THE ARKANSAS BLACKLAND ECOSYSTEM

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ABSTRACT

The North American Fire Learning Network (FLN), a joint project of The Nature Conservancy's Global Fire Initiative, the USDA Forest Service, and the U.S. Department of the Interior, was created in 2002 to catalyze efforts to reduce hazardous fuels across the United States. The primary goal of the FLN is to accelerate on-the-ground fire regime restoration across landscapes by fostering innovation and transferring lessons learned via regional learning networks. During its first 2 y, the FLN has engaged more than 250 partner agencies, tribes, and private landowners to advance ecologically sound restoration on more than 60 million acres. The south-central FLN is one of seven regional FLNs across the United States focused on collaborative restoration of landscape-scale fire-adapted ecosystems. One project within the south-central FLN is the Arkansas Blackland Ecosystem Restoration. The blackland ecosystem has been identified as one of the most at-risk ecosystems in the southeastern United States, with 600+ plant species and 315 animal species. A 2002 assessment of the region identified 35,000 acres in public and private ownership as targets for restoration to native blackland prairie and woodland. Currently, 6,000 acres are in conservation ownership. Restoration of the historic fire regime is crucial to restoring and maintaining the blackland prairie-woodland ecosystem. Through the FLN, The Nature Conservancy and the Arkansas Game and Fish Commission have developed restoration guidelines and management strategies for lands in various stages of degradation, current and desired future condition descriptions, and an ecological monitoring program to track restoration progress. The FLN has also helped in setting short-term and long-term priorities for the project and identifying funding sources for restoration activities. We present the products and accomplishments of the FLN in the blacklands of Arkansas. This presentation furthers the goal of the FLN by sharing successful implementation practices of a landscape-scale fire restoration project with ecologists, managers, and practitioners.

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SOUTHERN FIRE PORTAL: AN INTERNET-BASED PORTAL FOR FIRE SCIENCE AND MANAGEMENT IN THE SOUTHERN REGION

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ABSTRACT

The Southern Fire Portal (SFP), <http://frames.nbii.gov/southernfire>, aims to improve communication and collaboration between scientists and natural resource managers. The SFP proposes to achieve this goal by synthesizing vast, scattered, and difficult-to-locate fire-related information, and by providing efficient, free access to the syntheses on a single website. Envisioned as more than a website, SFP will be a gateway for ongoing information and technology transfer between the fire management and research communities, and their publics. Portal users can obtain, compare, and evaluate tools and information for wildland fire and fuels management. Users will be encouraged to give input on the performance and value of tools and information supplied through the portal. The southern region is geographically defined by the 13 states covered by the Southern Area Coordination Center (SACC), one of 11 geographic centers established to manage collaboratively wildland fire and other incident management activities. The SFP supports SACC by providing access to consolidated and organized fire research deliverables and other services necessary for effective fire and fuels management within this region. The SFP unifies several sources of fire information including 1) Fire Research and Management Exchange System; 2) Encyclopedia of Southern Fire Science; 3) Tall Timbers Research Station's E.V. Komarek Fire Ecology Database; 4) content being identified and collected from the Joint Fire Science Program and the National Fire Plan project by The Nature Conservancy Fire Network; 5) existing information through the U.S. Geological Survey's National Biological Information Infrastructure; and 6) key regional products and tools from state agencies, universities, and nongovernmental organizations. The SFP will provide single point access to fire-related publications, data sets, databases, decision-support tools, models, glossaries, interactive CD-ROMs, videos, and state-of-the-knowledge literature syntheses in support of fire and natural resource management for the southern United States. An advanced search engine and fire thesaurus enables users to quickly find the information they need.

Citation: Morgan, P., C. Fowler, D. Kennard, J.K. Hiers, R.E. Masters, J. Pollack, G. Gollberg, and A.M. Bruce. 2007. Southern Fire Portal: an Internet-based portal for fire science and management in the southern region [abstract]. Page 254 in R.E. Masters and K.E.M. Galley (eds.). Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida, USA.

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ABSTRACT

The Southeastern Fire Ecology Partnership is composed of fire ecologists from the U.S. Fish and Wildlife Service, USDA Forest Service, National Park Service, The Nature Conservancy, and Tall Timbers Research Station. The Southeastern Fire Ecology Partnership supports science-based ideas and innovations in land management issues related to fire. This information is widely communicated and accessible; it is extensively integrated into fire and land management decisions and actions, resulting in diverse, functioning landscapes. The partnership serves as a focal point for fire ecology information transfer, especially with regard to management issues on public lands and lands managed primarily for ecosystem health. The partnership's vision is that our efficient, creative, and cooperative efforts will lead to a workforce and society that understand and support the role of fire as a fundamental cultural, social, and ecological force. The mission of this group is to serve as a reference point for fire ecology issues in the southeastern United States and to support collaborative research, training, and education efforts within the field of fire ecology. As this partnership develops, it will serve as a centralized reference point known to academic, research, federal, state, and private communities regarding fire ecology issues in the Southeast. An additional objective is to serve as a clearinghouse for identification and prioritization of cooperative fire ecology-related research and educational needs. By capitalizing on co-location at Tall Timbers Research Station, we have a unique opportunity to pool resources and realize a synergistic effect based on differing perspectives from all partners.

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GOT TRUST? NATIONWIDE SURVEY RESULTS CONCERNING PUBLIC TRUST IN GOVERNMENT FUELS AND FIRE MANAGEMENT, AND PREFERENCES REGARDING FUELS REDUCTION METHODS

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ABSTRACT

Much of the public's attitude toward wildland fire as an important part of natural processes has been misguided, sometimes through programs perpetuating fear and misunderstanding of the vital role of fire in wildlands. Results presented here were part of a larger study that focused on the broad topic of public values, attitudes, and behaviors toward wildland fire. More specifically, the study was intended to contribute to development of a comprehensive understanding of public values, attitudes, and behaviors, and to understanding public preferences related to wildland fuels and fire management. Unlike previous research, this study aimed to provide national or "macro" level information. A broad-based national fire module was included with the ongoing National Survey on Recreation and the Environment (NSRE 2000). The survey is telephone-administered via CATI (computer-aided telephone interviewing) programs and random digit dialing through the Survey Research Center at the University of Tennessee, Knoxville. A stratified random sampling procedure was used, resulting in 6,979 completed responses. Results presented here will concern public trust in governments' abilities to manage fuels and fire, and public preferences concerning fuel reduction methods. Results indicate that while there is a general trust in land management officials there are still concerns, particularly regarding ability and capacity to manage for fire in forest and rangelands, use of taxpayers' money, and long-term forest health. The public also has some fairly clear preferences concerning fire management practices on public lands. Prescribed fire was most highly regarded between it, mechanical thinning, and chemical treatments. However, its use is not without concern. Harm to fish and wildlife, smoke management, and effects on scenery and recreation opportunities are among public concerns.

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