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Proceedings of a Special Workshop

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Editors:

W. Mark Ford, USDA Forest Service, Northeastern Research Station,
Box 404, Parsons WV 26287

Kevin R. Russell, Willamette Industries Inc.,
Box 488, Dallas, OR 97338

Christopher E. Moorman, Department of Forestry,
North Carolina State University,
Box 8003, Raleigh, NC 27695

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Preface

W. Mark Ford, Kevin R. Russell, and Christopher E. Moorman

Fire has a long history of regional use in the United States for forest, range and game management. Except for a few high-profile threatened, endangered, and sensitive species such as the pine barrens treefrog (*Hyla andersonii*), the red-cockaded woodpecker (*Picoides borealis*), and the Kirtland's warbler (*Dendroica kirtlandii*), the use of prescribed burning for nongame management efforts has been rare. Moreover, in areas such as the central hardwood regions, the Appalachians and the Pacific Northwest, silvicultural uses of fire have been discouraged, which consequently have limited applications of fire for nongame management. However, new emphases on biodiversity and community restoration are leading to re-examination of fire as a management tool in regions with fire-dependent systems, such as the southeastern Coastal Plain and in areas where fire historically has been excluded.

The timeliness and need of this symposium at the seventh annual meeting of The Wildlife Society was very apparent. As forest and land management on public lands continues to move from utilitarian or multiple-use approaches to more ecologically holistic management programs, prescribed burning is becoming an increasingly acceptable land management tool even where rarely used before. Conversely, on private lands, smoke management and liability issues and increased ownership fragmentation are having the opposite effect—decreasing the use of prescribed fire in land management activities. As natural resource scientists and land managers, not only must we gain a better understanding of the effects of natural and prescribed fire (or lack thereof) on nongame species, but also we need to promote its use when clearly indicated as a prescription for natural resource management. A critical component of this promotion is the education of the public by forestry and wildlife professionals about the historical role of fire and the importance of prescribed burning as a land management tool.

This symposium attempted to synthesize the widely scattered information and disparate databases that exist regarding the role of fire for managing nongame wildlife and relictual communities. Accordingly, this symposium drew upon the expertise of regional and nationally recognized wildlife scientists, resource managers, and fire ecologists with papers presented on the historical background, current issues, ongoing management efforts, and research and management needs of prescribed burning for both eastern and western landscapes.

Fire in the Eastern United States: Influence on Wildlife Habitat

D. H. Van Lear[1] and R. F. Harlow[2]

Abstract.—Fire is a major influence shaping wildlife habitats in the eastern United States. Lightning- and Indian-ignited fires burned frequently and extensively over the pre-Columbian landscape and shaped the character of numerous ecosystems. Depending upon the frequency, intensity, and severity of the fires, various assemblages of plants developed along environmental gradients, creating a shifting mosaic of habitats for wildlife. For millennia, fire was a major ecological process, mostly burning as frequent, light to moderate intensity surface fires in some ecosystems and as intense, stand-replacing fires in others. Within the past 100 years, fire has been excluded from most of the East and fire-maintained habitats have dramatically declined. In many cases, wildlife species dependent on these habitats are in decline or experiencing dangerously low numbers. Unfortunately, the trend toward a relaxation of bans on burning may soon be reversed due to the adverse publicity from recent disastrous fire in the West.

Wildfires and escaped prescribed fires frequently have been in the news in recent years, e.g., those in Yellowstone National Park in 1988, Florida in 1998, and throughout the West in 2000. Usually, only catastrophic wildfires or escaped prescribed fires are deemed newsworthy. Media coverage often concludes that forests have been destroyed by these wildfires and imply that wildlife habitat has been destroyed as well. But has wildlife habitat really been destroyed? Does fire so damage ecosystems that they are gone forever? Of course not, especially in the eastern United States where vegetative recovery is relatively rapid following disturbance. From an ecological perspective, ecosystems and their associated wildlife habitats are not destroyed, only changed to an earlier stage of succession.

Fire should be considered in the context of its role as a natural ecological process that has shaped vegetative patterns in most eastern ecosystems for millennia. Even ecosystems burned with high intensity fires generally have mechanisms in place to recapture solar energy and recycle nutrients, allowing rapid ecosystem renewal. All over the world, early man used fire for his benefit. Properly used, it remains an important tool in both forest and wildlife management.

The earth has been described as a fire planet, a planet whose terrain is covered with combustible organic fuels which continually accumulate over time, an oxygenated atmosphere that supports combustion, and two potent ignition sources–lightning and man (Pyne et al. 1996). Most ecosystems on earth have burned in the past and will burn again in the future–creating, changing, or maintaining habitats for a wide variety of wildlife. Because fire was historically so prevalent and frequent across the eastern landscape, wildlife evolved adaptations to fire in order to survive (Komarek 1974, Brennan et al. 1998).

Fire History in the Eastern United States

Approximately 12,000 years ago, man first made his way into North America (Williams 1989, Bonnicksen 2000). Within a relatively short period of less than 1000 years, these early Americans had occupied nearly all of the western hemisphere. They used fire in many ways because it helped them survive. Over thousands of years, the American Indian became expert in using fire for various purposes, e.g., for hunting, to concentrate prey species in convenient areas, to encourage fruit and berry production, to keep the woods open along major corridors of travel, to fire-proof their villages, and other uses (Williams 1989, Pyne et al. 1996, Bonnicksen 2000). Through trial and error over millennia, the Indian learned to make fire his servant and not his master. Thus, controlled, or prescribed, burning is nothing new (Komarek 1974).

The Indian population of North America at the time of Columbus was estimated by Dobyns (1983) at about 18 million. This figure has been widely quoted, although Stannard (1992) more recently suggested that there may have been as many as 100 million Indians in North America. Regardless of the accuracy of these and other estimates of the Indian population, there can be no doubt that, at the time of Columbus, there were millions of Indians modifying the "natural" forests. Their influence on the eastern forest was widespread because of their use of fire and extensive land clearing for agriculture (Hudson 1976, Williams 1989). Because of their farming and burning activities, Indians ensured that much of the eastern forests was in early successional habitats.

Indians often burned as frequently as twice a year, complementing lightning as an ignition source. Their burning extended the fire season beyond the "natural" lightning-fire season of summer. These frequent and often extensive fires, along with the wildlife foraging that fire encouraged, created and maintained open woodlands, savannahs, and prairies throughout the eastern United States (Williams 1989, McCleery 1993, Kay 2000). In fact, much of the eastern forests at the time of Columbus could be regarded as a cultural artifact of Indian activities.

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[1] Department of Forest Resources, Clemson University, Clemson, SC 29634

[2] Deceased; Southern Research Station, USDA Forest Service, Clemson, SC 29634

Because of burning and agriculture, the eastern United States at the time of Columbus was no "forest primeval" composed of closed canopy, old-growth forests. The oft-repeated phrase that a squirrel (*Sciurus* sp.) could travel from the Atlantic coast to the Mississippi River in pre-Columbian times without ever touching the ground is patently false (Buckner 1983, Hamel and Buckner 1998). The eastern forest at that time was a shifting mosaic of woodlands, savannahs, forests, and prairies, all in varying stages of succession. After Indian populations plummeted in the 16th century from exposure to European diseases, their level of burning declined and prairies gradually became woodlands, savannahs became dense forests, and dense undergrowth invaded open forests (McCleery 1993).

The early European settlers found that much of the East was already being fired on a frequent basis by Indians (Komarek 1974). These settlers gradually displaced the Indians, but continued their use of fire for many of the same reasons, i.e., to clear the woods of underbrush, to expose nuts, to clear agricultural fields, etc. Frequent fires occurred over large areas of the eastern landscape into the early decades of the 20th century.

Not until the early 1900s were there serious efforts to exclude fire as an ecological process in eastern North America. In 1910, an outbreak of wildfires in the western United States--concentrated in Idaho and Montana, burned millions of acres, much of which was on national forests, and killed 78 firefighters (Pyne et al. 1996). These events occurred just after the Forest Service had made fire control a top priority on the nation's national forests and caused the public to become concerned about the sustainability of the nation's natural resources. Led by the Forest Service and the state forestry commissions, the public began to see fire as an enemy to be suppressed at all costs. An era of fire suppression began that created different environments from those that had existed for millennia, often to the detriment of many wildlife species.

Habitats and Fire

Habitats are environments that provide the necessities of life for wildlife, i.e., food, water, cover, and places to rear their young, and are shaped by ecological processes acting on varying gradients of soil, topography, hydrology, and climate over time. In most of the eastern United States and in many other parts of the world, fire historically played a major role in shaping habitats. Fire, often working in combination with other disturbance factors, was the major factor determining the composition, structure, and pattern of vegetation that occupied different portions of the landscape.

Fire became a more important force in shaping habitats in the East as the climate warmed following the last glaciation because a warmer climate promoted better burning conditions. Indian populations were also increasing, especially in the past few thousand years, as greater use of agriculture allowed them to discard their previous nomadic lifestyle. And wherever there were Indians, there was fire.

The Laurentide ice sheet, which covered the northeastern United States and extended south to the Ohio River at the peak of the Wisconsin glaciation, began to retreat about 18,000 years ago (Bonnicksen 2000). At the end of this ice age about 10,000 years ago, vegetation in the eastern United States was still migrating northward (Delcourt and Delcourt 1987). It is difficult, if not impossible, to unravel effects of a changing climate (generally warming until about 6,000 years ago, then generally cooling again–until about 1970) from the effects of Indian burning. By about 5-6,000 years ago, the vegetative associations of today were essentially in place in the South, although the structure of those forests must have been much different than today's forests. In the Northeast and Midwest, it took longer for individual species to arrive in their current locations because they had farther to travel from their southern refugia.

Burning altered the structure, composition, and pattern of vegetation. Fire regimes created types of vegetation that actually encouraged recurring fire, e.g., the longleaf pine (*Pinus palustris*)-wiregrass (*Aristida*) ecosystem, and, in so doing, shaped habitats. The habitats that developed depended upon the type of fire regime, the adaptations of plants occupying the landscape, and the length of time since the last disturbance. The magnitude of the historical role of fire is under-appreciated by the general public and even some resource managers because of almost a century of efforts to exclude it from the majority of the landscape.

Historical Fire Regimes

In the eastern United States, some areas were burned so infrequently, e.g., moist to wet areas in northern hardwood forests, that it can be said that these areas do not burn. Other areas were subject to annual fires that burned with low intensity, i.e., low flame lengths, and low severity, i.e, little damage to soil or overstory vegetation. Still other areas burned infrequently but with great intensity and killed essentially all overstory trees. Because of the many types of fires that can occur, fire scientists have devised systems for characterizing them (Table 1), using variables such as fire intensity, severity, and frequency of burning to describe fire regimes (e.g., Heinselman 1981).

Understory fire regimes are characterized by frequent burns of relatively low intensity that do little damage to soil or overstory trees (low severity). Ecosystems subjected to frequent understory burns recover quickly as the grasses, forbs, and understory woody vegetation characteristic of this type of fire regime resprout vigorously within weeks after burning. Overstory trees

Table 1.—Heinselman's (1981) classification of historic fire regimes, slightly modified, with selected eastern examples. (Note: Only those classes applicable to eastern conditions are given.)

Class	Characteristics of Fire Regime	Eastern Examples
0	No natural fires	Wetter regions of the eastern deciduous forest
1	Infrequent, light surface fires with more than 25-yr return intervals	Many eastern deciduous forests
2	Frequent, light to moderate surface fires with 1- to 25-yr return interval	Southern pine forests of Coastal Plain and lower Piedmont; Tall-grass prairies
3	Infrequent, severe surface fires with more than a 25-yr return interval	Dry jack pine-oak "barrens" of Minnesota, Wisconsin, and Michigan
4	Short return interval crown fires and severe surface fires in combination (25- to 100- year return intervals)	Most boreal forests in eastern Canada and United States; Sand pine

are generally little affected by these low intensity fires. Such a fire regime was typical of the longleaf pine-wiregrass ecosystem that dominated up to 90 million acres of the Coastal Plain of the Southeast at the time of Columbus, but which now occupies only about 3% of its original range (Frost 1998, Landers et al. 1995). As a result of loss of this open woodland, savannah type habitat, numerous wildlife species including the red-cockaded woodpecker (*Picoides borealis*), gopher tortoise (*Gopherus polyphmus*), and southern fox squirrel (*Sciurus niger*) are in decline.

Stand-replacement fire regimes are typified by infrequent fires of high intensity that kill most of the overstory trees and sometimes damage the soil (high severity). Such fire regimes are typical of sand pine (*Pinus clausa*) stands in Florida (Outcalt and Balmer 1983), Table Mountain (*Pinus pungens*)/pitch pine (*Pinus rigida*) stands in the southern Appalachians (Clinton et al. 1993), and jack pine (*Pinus banksiana*) stands in the boreal forests of the northern United States and Canada (Rudolph 1983, Bonnicksen 2000). Fires in these types may burn as surface fires under mild conditions but under severe weather and fuel conditions stand-replacement fires develop. In the absence of fire, especially on better sites, the species are replaced by more shade-tolerant species, i.e., scrub oaks in sand pine (Outcalt and Balmer 1983), oaks and mountain-laurel (*Kalmia latifolia*) in Table Mountain/pitch pine stands (Waldrop and Brose 1999), and red pine (*Pinus resinosa*), white pine (*Pinus strobus*), and various hardwoods in jack pine stands (Rudolph 1983). Following stand-replacement type fires, seeds from serotinous cones germinate on a fire-prepared seedbed and a new stand develops. However, ecosystem recovery may take decades before the character of the new stand recovers to that of the pre-burned stand. Burning in these types requires intensive, experienced supervision because of the high risk of fires getting out of control.

The jack pine ecosystem is one of the most extensive in North America and exists in a variety of successional stages on many site types. Jack pine grows best on dry soils where fires are relatively frequent, i.e., every 50 - 100 years, and severe (Bonnicksen 2000). There are many wildlife species that are characteristic of these habitats, although we know of few that are threatened or endangered. Many species are cyclic in the jack pine ecosystem, including the snowshoe hare (*Lepus americanus*), Canada lynx (*Lynx canadensis*), and the snowy owl (*Nyctea scandiaca*). The Kirtland's warbler (*Dendroica kirtlandii*), one of America's rarest birds, nests only in pure stands of jack pine (Hunter 1990).

The Table Mountain/pitch pine ecosystem in the southern Appalachians is in decline because of fire exclusion in this type. There is some debate among fire scientists about whether the Table Mountain/pitch pine type in the southern Appalachians was maintained primarily by stand-replacement fires or periodic understory fires. Recent research by the North Carolina Wildlife Resources Commission, the U.S. Forest Service, and Clemson University suggests that periodic burning in the type creates open woodland conditions, which would enhance landscape diversity within the sea of mature, dense hardwood and pine forests that now dominate the southern Appalachian mountains.

Not all stand-replacement fires occur in forests. The tallgrass prairie was dominated by fire-tolerant tall grasses, including big bluestem (*Andropogon* sp.), little bluestem (*Schizachyrium scoparium*), and Indiangrass (*Sorghastrum nutans*), but also contained forbs, shrubs, and trees. This ecosystem, which existed in pure form in

Table 2.—Plant adaptations to fire in different stages of development.

Stage of Development	Adaptation	Plant Examples
Vegetative	Sprouting from root collar buds	Most hardwoods; certain pines
	Sprouting from adventitious or latent axillary buds	Most southern pines
	Thick bark	Most oaks; southern yellow pines; numerous others
	Protected buds on boles and limbs	Pitch pine and pond pine
	Protected buds below ground	Oaks
Reproductive	Precocious flowering	Table Mountain pine; Sand pine; jack pine
	Stimulation of flowering	Wire grass
	Serotinous cones	Table Mountain pine; pitch pine; jack pine; sand pine
Germination	Heat-enhanced seed germination	Some legumes; fire cherry; yellow-poplar
	Germination on fire-prepared seedbeds	Many pines; fireweed; many other species
Other Adaptations	Increased inflammability	Longleaf pine, wiregrass, jack pine

eastern Nebraska, Illinois, Iowa, and eastern Kansas, graded into a forest-prairie complex to the east, and has been almost totally converted to agriculture (Telfer 2000). Only about 1% of the tallgrass prairie remains. It was a fire-maintained ecosystem that was burned primarily by man at intervals of 1-3 years (Pyne et al. 1996). When fires became less frequent, the eastern portions of the prairie succeeded to shrubs and trees. When burned annually, shrub species were eliminated. Wildlife species in decline because of this vanishing habitat include the greater prairie chicken (*Tympanuchus cupido*), badger (*Taxidea taxus*), and false map turtle (*Graptemys pseudogeographica*).

Some fire regimes may include both surface fires and stand-replacement fires. There are mixed severity fire regimes where fires alternate between understory burns and stand replacement burns, either within the same fire or in different fires. Such burns cause selective mortality in the overstory, especially when certain species are better adapted than others to withstand the effects of high intensity fires. Burns of this type occur, for example, in mixed pine/hardwood stands in the southern Appalachians where fuel loadings have accumulated primarily in the form of dense understories of mountain-laurel (Elliott et al. 1998, Waldrop and Brose 1998). Fires in this type are intense enough to topkill many of the hardwoods and shrub understory but the thick bark, high crowns, and ability to quickly regenerate new needles allows many of the mature pines to survive.

Fire Adaptations

Fire is only one of many evolutionary forces that shape plants and plant communities. However, fire has been so ubiquitous on the planet and exerts such profound influences on the environment that it is reasonable to assume that fire regimes selected for individuals in populations that could best survive fire (Pyne et al. 1996). Plant species that evolved with fire adapted by developing characteristics that increase their chances of survival in various types of fire regimes (Table 2). Such adaptations occur in various stages of a plant's development, from regeneration to maturity (Kimmins 1987).

The ability to resprout from root collar buds following topkill by fire is probably a fire adaptation in some species, even though this characteristic occurs in other taxa where fire is alien (Bond and van Wilgen 1996). Many species replace scorched foliage by sprouting from adventitious or latent axillary buds on branches or boles of trees. Where fire is frequent, sprouting adaptations such as these improve the survival potential of plants because it allows those plants to continually occupy the same growing space. Non-sprouters would be favored by longer return intervals between fires because they could allocate more of their energy to top growth, allowing them to more completely occupy the growing space in the absence of fire for long periods.

Some species have developed thick bark apparently to insulate the living cambium from the heat of surface fires. Other species have thin bark when young and are quite susceptible to fire damage or mortality, but develop very thick bark as they mature and become resistant to surface fires. The protection offered by bark is dependent upon not only its thickness but, at least for some species, by its thermal properties (Hare 1965).

Certain fire-adaptations are even more species specific. Cone serotiny, where seed accumulates in unopened cones that open after being heated by fire, is characteristic of relatively few species in the East. Species with serotinous cones normally exist in environments where high intensity, stand-replacing fires are the norm (Pyne et al. 1996, Bond and van Wilgen 1996). Another species-specific adaptation is the grass stage of longleaf pine, where their fire-sensitive buds are protected from overheating by a dense sheath of moisture-rich needles (Walhenberg 1964). This characteristic is unique among the pines.

Oaks in some ways might be considered to be the longleaf pine of the hardwoods. They have multiple adaptations to fire, including thick bark that protects them from damage by surface fires. Oaks also have hypogeal germination, i.e., their root collar buds are below the surface of the soil because their acorns are buried by squirrels and/or jays (*Sciuridae* and *Corvidae*). Because these buds are protected from the heat of surface fires (soil is a poor conductor of heat), oaks are better able to successfully sprout following fire than many of their competitors whose seeds germinate on the soil surface, such as yellow-poplar (*Liriodendron tulipifera*) (Brose and Van Lear 1998, Brose et al. 1999).

Many species in the eastern United States have seeds that are either encouraged to germinate by heating or to germinate on fire-prepared seedbeds (Pyne et al. 1996, Lyon et al. 2000). Heat from fire can crack the seedcoat and release the seed from dormancy. Some grasses, such as the wiregrasses, only flower when burned in certain seasons, with blooming following spring burns but not after winter fires (Brennan et al. 1998).

Some adaptations not only allow plants to survive fire but also play a major role in predisposing plant communities to recurrent fire (Mutch 1970). For example, the dominance of wiregrass in the herbaceous layer, with its high proportion of dead leaves and their overlapping nature, along with the resinous nature of fallen longleaf pine needles, insures that surface fires will be frequent in this ecosystem (Landers et al. 1995). Another example would be in oak forests where the fallen leaves curl and resist decay, allowing a highly aerated fuel bed to develop, which encourages frequent surface fires. In contrast, leaves of fire-sensitive hardwoods like American beech (*Fagus grandifolia*), yellow-poplar, and red maple (*Acer rubrum*) lie flat on the ground and decay rapidly, preventing a flammable fuel bed from developing. Unless fire-adapted plant communities tend to promote recurrent fire, they are likely to be replaced by non-flammable communities (Bond and van Wilgen 1996).

These adaptations of different plant species, along with conditions for establishment and the availability of plant propagules, determine patterns of succession and the types of habitats that develop following burning.

Effects of Fire on Habitat Components

Fire affects habitat components in many ways. Periodic understory burns cause many plants to resprout and these new sprouts, whether woody or herbaceous, are more palatable and have higher protein and nutritive content than the older tissues that they replaced (Stransky and Harlow 1981, MacCracken and Viereck 1990). Enhanced palatability and higher nutrient contents in post-burn growth are relatively short lived (generally 1-2 years), but are important in areas frequently burned. Regrowth after burning is not only more palatable and nutritious than older tissue, it is also more available because most sprouting occurs at ground level (Lay 1967).

Frequent burning tends to favor herbaceous vegetation over woody vegetation, i.e., it increases the proportion of herbage to browse (Lewis and Harshbarger 1976, Pack et al. 1988). Growing season burns discriminate against woody growth more than dormant season burns. Frequent burning creates open forest conditions in both pine and hardwood forests, i.e., understory and midstory woody vegetation are replaced by herbaceous vegetation (Landers et al. 1995, Thor and Nichols 1974). Burning does not eliminate woody growth but controls its size (Waldrop et al. 1987). Annual summer burns in southern pine stands will eventually remove hardwoods from the stand but it takes numerous burns. Certain woody species, such as oaks, tend to be more tenacious sprouters than competitors and therefore persist in frequently burned ecosystems.

Seed and mast production generally increase after burning in Southern forests. Seed production by legumes, grasses and spurges is usually greater on frequently burned areas (Cushwa et al. 1970, Robbins and Myers 1992). Production of berries, drupes, and pomes normally peaks about 2 to 4 years following burning (Harlow and Van Lear 1989). However, burning too frequently, i.e., annually or biennially, reduces most mast-producing species (Robbins and Myers 1992).

Numerous hard-seeded species require their seed to be heated to 60 - 80°C to break dormancy (Auld 1986). Some shrub species (*Vaccinium*) and small trees such as dogwood (*Cornus florida*) tend to be more fruitful a few years following burning, although others do not (Harlow and Van Lear 1989). Hard mast-producing species, such as oaks and hickories (*Carya* sp.), are generally favored by fire during the regeneration period (Abrams 1992, Barnes and Van Lear 1998, Brose et al. 1999).

Effects of fire on standing dead trees, i.e., snags, and fallen woody debris are variable (Lanham and Guynn 1993, Van Lear 1993). Surface fires burning under droughty conditions frequently kill stressed trees, creating snags. In addition, fires may scar the bases of some trees and eventually become basal cavities. However, some snags may be consumed by fire or at least brought to the ground, where they become fallen woody debris. Both forms of coarse woody debris are extremely important habitat components (McMinn and Crossley 1993). In the East, most snags do not stand for long periods of time, often falling within a decade of death (Dickenson et al. 1983, Sabin 1991). Coarse woody debris is generally not consumed by fire unless the drought index, e.g., the Keetch-Byrum drought index, is high. Only when the debris becomes dry throughout during droughty conditions will significant amounts be consumed by fire.

In the southeastern United States, long-term studies in the Coastal Plain show that periodic burning increases availability of phosphorus, as well as other nutrients in the mineral soil, probably because of increased mineralization and because soil pH is raised (McKee 1992). Most soil nutrients become more available for plant uptake as soil acidity approaches neutral. This increased availability probably accounts, at least in part, for the higher nutrient levels in regrowth following fires. Enhanced soil phosphorus levels resulting from burning may have implications in Quality Deer Management programs because phosphorus is needed for antler development (Cowan and Long 1962).

Burning historically provided a mosaic of habitat types in a variety of seral stages and fuel conditions over the landscape (Wood 1981, Williams 1989, Pyne et al. 1996). Fire did not burn uniformly across the landscape, but interacted with topographic, edaphic, vegetative, and weather conditions to produce a mix of habitats and ecotones. While fire occasionally changed mature forests to early successional habitats, i.e., jack pine ecosystems, it also maintained the structure and composition of certain ecosystems where fire was a frequent visitor, e.g., the longleaf pine-wiregrass ecosystem and the tallgrass prairie ecosystem. These varied conditions created habitats for many species of wildlife which, over millennia, came to depend on these conditions. As the role of fire has declined over the past century, so have many plant and wildlife species.

Conclusions

Fire has shaped the pattern, composition, and structure of vegetation in the eastern United States for millennia, especially since the arrival of Indians at least 12,000 years ago. Indians used fire for many purposes and often fired the woods on an annual basis. Frequent burning kept the forests more open and increased the ratio of herbaceous/woody vegetation. Fire did not burn uniformly across the landscape but created a shifting mosaic of habitats depending upon fuel, weather, and site conditions at the time of burning. In some ecosystems, fire burned frequently but with low intensity and severity and maintained the *status quo* of the ecosystem. In other cases, fires burned with high intensity and severity and completely changed the ecosystem. Plants and animals adapted to these different fire regimes in order to survive. Over time, many wildlife species became dependent upon these fire-maintained habitats.

During most of the past century, attempts have been made to exclude fire from much of the eastern landscape. Exclusion of fire, along with other factors, have resulted in major declines in fire-maintained habitats, the most notable examples being the longleaf pine-wiregrass habitat and the tallgrass prairie habitat. Fire also played a role in maintaining other habitats important to wildlife. Historically, it perpetuated the jack pine, sand pine, and Table Mountain/pitch pine ecosystems and was instrumental in maintaining oak forests on good-quality sites throughout much of the East. The exclusion of fire in fire-maintained habitat types may make catastrophic wildfires more frequent, potentially changing ecosystems and eliminating desirable habitat for some species. Burning on a regular basis prevents fuel build-up and minimizes the chances of a catastrophic wildfire occurring.

There is no question of the historical importance of fire in creating and maintaining wildlife habitats in the eastern United States. However, we now live in a different time and there are numerous restrictions on the application of fire for wildlife habitat purposes. We suggest that prescribed fire be used where feasible and where it can accomplish desired objectives, because it has unique habitat effects. However, in the future it will be necessary to use other management techniques, including harvesting, thinning, and herbicides, to mimic, at least partially, the effects of prescribed fire on wildlife habitat where burning is not appropriate.

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The Historical Foundations of Prescribed Burning for Wildlife: a Southeastern Perspective

A. Sydney Johnson[1] *and Philip E. Hale*[1]

Abstract.—Controlled burning has deep historical roots in the South, where the practice was quickly adopted from the Indians by early European settlers. It became used widely, primarily to improve forage conditions for free-ranging cattle and to improve visibility and access. Likewise, hunting is deeply imbedded in southern culture and was an attraction to visitors throughout the 19th Century. This was especially true of quail (*Colinus virginianus*) hunting, and after the Civil War wealthy northerners began to buy large plantations for hunting retreats. In the 1920's Herbert L. Stoddard documented the necessity of prescribed burning to maintain bobwhite quail habitat on these plantations. Opposition to the practice among foresters and public agencies was fierce, and Stoddard became an outspoken advocate of light winter burning in longleaf pine (*Pinus palustris*) and, later, certain other forest types. Use of prescribed fire in forestry and game management was gradually accepted. But, although some naturalists such as Stoddard were interested in the effects of fire on native flora and nongame wildlife, private landowners and public agencies generally showed little interest in managing specifically for non-game wildlife until the 1970's. By then, there was in the southern states a background of 50 years of research and many more years of practical experience in the use of fire that could be applied to this new goal. Soon, any biologists and managers recognized that prescribed burning would play a nearly essential role in managing certain nongame species. And, as new management goals evolved, fire regimes other than light winter burning also came under scrutiny for potential use in restoration and maintenance of certain natural communities

Introduction

Woods burning has a long unbroken history in parts of the South, and much of the early scientific work on prescribed burning was done there. In the keynote address to attendees of a prescribed burning symposium, E. M. Bacon (1971) of the U.S. Forest Service said "Prescribed burning seemingly had its origin in the South, it has certainly been an accepted management tool for a longer period of time in this region than anywhere else in the country." In his book "Fire in America," Pyne (1982) noted that when the need for prescribed burning became evident, the skills were in the hands of local woods burners, "a resource that had to be managed." He concluded that "Its peculiar fire heritage helped the South to train the rest of the nation in the art of prescribed burning."

There is too much literature on the subject to review in detail. The history of the controversy surrounding controlled burning and the forestry profession has been dealt with in detail by others. There are at least two important books dealing with the subject (Pyne 1982; Schiff 1962). Our objective is to provide a historical overview of the southern roots of prescribed burning in forestry and, especially, wildlife management. We will mostly limit our discussion to the application of prescribed burning in management and will not review research on habitat and wildlife responses to fire. The term prescribed burning originated among foresters and came into use in the 1940's. The term controlled burning was used by the local people and, until recently, by wildlife managers. Although the two terms have somewhat different definitions, we will use them interchangeably, as appropriate in the context of the discussion.

Prehistoric Fires

Fire is more common and more important in the environment of the South than in most other areas of the United States (Christensen 1978). This is especially true in the Coastal Plain, but even in the Appalachians fire has been frequent enough to cause the development of fire-dependent species and influence the composition of forests over large areas (Van Lear and Waldrop 1989). The frequency, seasonality and intensity of its occurrence are determined in part by local climate, topography, soils, and vegetation. Plants adapted to specific sets of conditions are sorted into communities that may have characteristics that facilitate or deter fire, or influence the type of fire regime. Periodic fire is especially an important part of the environment of all the southern (yellow) pines (*Pinus* spp.) except spruce pine (*P. glabra*), which typically grows in mixed stands with hardwoods. Landers (1991) stated "the pine genus has long been recognized as having special traits adaptive to fire and fire-prone sites," and he provided a detailed analysis of these traits and their relationship to fire periodicity and intensity for each of nine southern pine species. Today, nearly half of the total forest area in the southern states and nearly two-thirds of that in the Coastal Plain and Piedmont is occupied by pine or pine-hardwood types, (estimated from data in Rudis 1998).

But pine forests are not the only fire-adapted communities. Other community types that are dependent upon periodic fires for their regeneration or

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[1] Daniel B. Warnell School of Forest Resources, University of Georgia, Athens, GA 30602

maintenance include shrub bogs, Atlantic white-cedar (*Chamaecyparis thyoides*) swamp forests, bay forests, cypress (*Taxodium ascendens*) domes, and grass-sedge bogs in the Coastal Plain (Christensen 1978, 1988; Garren 1943; Komarek 1974,), heath thickets and grassy meadows at high elevations in the Southern Appalachians (Christensen 1978; Garren 1943), and canebrakes along stream bottoms in all provinces (Hughes 1966; Komarek 1974; Platt and Brantley 1997).

Lightning is assumed to have been the primary agent that caused fires in North America before humans became a major factor (Komarek 1964, 1966; Pyne 1982). Until recent centuries there were few natural barriers to the spread of such fires after ignition, and fires probably swept unimpeded over vast areas. However, some (e.g., Stewart 1956) have questioned whether lightning played the primary role in shaping pine forests in much of the South. They have noted that lightning in the South is usually accompanied by rain, and lightning-caused fires in the region today are usually spotty and small in size compared to man-caused fires (Barden and Woods 1976).

Anthropologists and geographers tend to emphasize man as the main source of prehistoric fire (Denevan 1992; Stewart 1956). Paleoecological studies (Delcourt and Delcourt 1985; Delcourt et al. 1993; Whitehead and Sheehan 1985) indicate there was a rapid expansion of southern pine forests 3,000-6,000 years ago. This occurred concurrently with the expansion of American Indian populations and may have been at least partly due to their actions. But, it is impossible to sort out the effects of climatic changes that were occurring during that time. Many writers have documented deliberate use of broadcast burning by the Indians (DeVivo 1990; Hammett 1992; Maxwell 1910; Robbins and Myers 1992; Rostlund 1957; Silver 1990; Stewart 1956; Van Lear and Waldrop 1989). Spanish explorers and, later, other travelers consistently described an open landscape and gave eyewitness accounts of Indians using fire in clearing land for agriculture, driving game, and other purposes. Rostlund (1957) reviewed many of the early narratives and concluded that Indians were burning the landscape and creating savannas and prairies at an accelerating rate until European contact in the 16th Century. Increasing amounts of charcoal in deposits laid down over the last 1,000 years (Delcourt and Delcourt 1985, 1997) support Rostlund's conclusion.

After contact with Europeans, Indian populations were severely decimated by disease and other factors—possibly being reduced by as much as 90 percent (Lovell 1992). The prairies and savannas returned to forest, and the extensive pine forests described by so many travelers in the 18th and 19th centuries (Johnson 1987) probably developed after the influence of the Indians was greatly diminished, or removed entirely. Regardless of the relative importance of man versus lightning as a pyrogenic agent shaping vegetation, it is evident that for several thousand years, fires set by Indians were significantly affecting the landscape (Delcourt and Delcourt 1997).

The Tradition of Woods Burning in the South

As European settlers moved in, fire was used to clear land for farms and communities; and, nearly everywhere on the frontier, fire was used carelessly at first. But, controlled burning did not become a common practice in the Northeast. Much of the region was mesic hardwood forest (Kingsley 1985), not a fire type, and there were fire control laws in all of the Northeast by the time of the Revolution (Pyne 1982). Commercial logging in the region began early, and timber interests encouraged fire suppression (Pyne 1982). Also, the Northeast was settled mainly by people from the southeastern lowlands of England (Fischer 1989), and, later, from parts of continental Europe where forests were predominantly fire-sensitive hardwoods or spruce (*Picea* spp.). Immigrants from these areas had little experience with controlled burning (Pyne 1982). They came from some of the most heavily urbanized areas of the time; few of them had lived on farms (Fischer 1989), and most that had lived on farms came from areas where row crops and fenced pastures were the dominant pattern of agriculture (Fischer 1989; McWhiney 1988; Pyne 1982). They developed agricultural patterns in America similar to those with which they were familiar (Fischer 1989).

But, in the South, woods burning was a widespread practice from the outset, especially in the Coastal Plain. As previously shown, much of the region supported fire type forests. Settlement was primarily by immigrants from the uplands of rural western England, and later from Scotland and Ireland (McWhiney 1988, Fischer 1989), where open range herding was customary (McDonald and McWhiney 1975; McWhiney 1988; McWhiney and McDonald 1985). Many of the settlers probably were already familiar with the practice of controlled burning to improve grazing conditions. The forests in the highlands of England, Scotland and Ireland had been converted to heathlands by centuries of burning for grazing by cattle and, later, sheep and red grouse (Gimingham 1970; Kayll 1966). In much of the South people called "crackers" (sometimes disparagingly) practiced a frontier lifestyle centering around free-ranging livestock (cattle and hogs), patch farming, dipping turpentine, and hunting and fishing (McWhiney 1988; Owsley 1949). Pyne (1982) noted that "for many of the Scotch-Irish immigrants who settled the regions, the socio-economic environment was not unlike that of Scotland, which had helped to perpetuate a herding and hunting economy that routinely used broadcast fire." McWhiney and McDonald (1985) observed "The custom of range burning. . .seems to have been another adaptation in

America of a Celtic tradition—one that meshed with an American Indian practice.″ McWhiney (1988) stated that open range herding of livestock was “a continuation in the Old South of traditions practiced for centuries by Celtic peoples.″ In Florida and the Southwest, there was a similar Spanish influence.

As settlement continued, the better lands, especially in the Upper Coastal Plain, Piedmont and Valley provinces, were increasingly occupied by descendants of English Cavaliers who had moved inland from coastal Virginia and the Carolinas. They were mainly farmers and planters, who cleared the forests, cultivated the land and developed permanent lifestyles (Owsley 1949). Here, the plantation culture was best developed and, because most of the land was cleared for row crops and improved pasture, in many places the tradition of burning the woods was broken. Those who were feeling crowded and unwilling to change their ways and become farmers, moved to less productive lands (Owsley 1949) or on to the West (Jordan 1981). They were able to pursue their traditional ways in the extensive pinewoods of the Coastal Plain, the Southern Appalachians, and scattered areas throughout (Owsley 1949). In these areas woodlands were treated as a commons for hunting and grazing.

Popular interest in the Old South has focused mainly on the large cotton and rice plantations, while, until recently, the other livestock producers have gone unrecognized except by professional historians. Unlike the plantation owners, small farmers and others who lived off the land left little in the way of written records. McWhiney (1988) cited data showing that for 15 years before the Civil War the average number of livestock driven to market from the South each year was many times the annual numbers in the famous Texas cattle drives during their heyday for 15 years after the war. Just before the Civil War, the cash value of southern livestock equaled the cash value of cotton and all other crops combined (McWhiney 1988). Much (but not all) of the livestock was produced on open range. Fencing of one’s property was illegal in some southern states until the middle 1800’s (McDonald and McWhiney 1975); and, in most states of the Deep South, unrestricted ranging of livestock was allowed under local option laws until after World War II. It was up to the landowner to fence livestock out, and motorists were liable for damages in accidents involving livestock on roads and highways.

Many of the herdsmen tended large herds of cattle and hogs but owned little or no land. Some were cowboys hired by absentee livestock owners. In the pinewoods large herds were tended from horseback, and controlled burning opened up the landscape, allowing better access and visibility. Other reasons for burning included reducing the hazard of wild fire to turpentine woods, reducing the risk of rattlesnake (*Crotalus* spp.) bite, and controlling ticks. Often no attempt was made to confine fires within property boundaries, nor was there a desire on the part of most landowners to exclude fire from their property. Controlled burning was often a community affair.

Fire also was used as a tool in hunting, however, did not find any references documenting prescribed burning specifically to improve game habitat. Nonetheless, hunting was an integral part of Southern culture among all social and economic classes (Elliott 1846; Gohdes 1967; Marks 1991; Oppel and Meisel 1987). Skilled hunters would have known which habitat conditions provided best hunting success and what practices resulted in those conditions. And they almost certainly would have used these practices to increase hunting success—especially on the plantations, where sportsmen from the northern states and Europe were entertained throughout the nineteenth century.

The Quail Plantations and Controlled Burning

Hunting in the South, especially for bobwhite quail (*Colinus virginianus*) hunting, became increasingly attractive to northern visitors after the Civil War. As land became available during the economically difficult times following the Reconstruction era, wealthy northerners began to buy plantations for hunting retreats in South Carolina, Florida, southwestern Georgia, and, later, in other areas of the Southeast (Brueckheimer 1979; Paisley 1968). One of these owners, Henry L. Beadel, told of how he and other new owners brought northern attitudes about fire with them and were appalled at the extensive burning (Beadel 1962). They put a stop to it on their properties, and as a woody understory developed, quail populations declined.

But, these plantation owners were educated and influential. In 1923 they began discussions with the U. S. Bureau of Biological Survey (forerunner of the Fish and Wildlife Service) that led to a cooperative study of the bobwhite quail and the reasons for its decline. Herbert L. Stoddard was employed by the Bureau to head the project, which was financed by the plantation owners. The final report on the study was published in book form and remains a classic in wildlife literature (Stoddard 1931). In it, Stoddard identified lack of fire as a cause of quail decline. He became an outspoken advocate for the use of light winter fires in game management and forestry, especially in the longleaf pine (*P. palustris*) type. After completion of the original quail investigation., Stoddard continued to promote prescribed burning as director of the privately funded Cooperative Quail Study Association from 1931 to 1943 and, after that, as a forestry and wildlife consultant, and still later, as one of the founders of Tall Timbers Research Station near Tallahassee, Florida.

The Controlled Burning Controversy

For years a conflict had been developing between timber interests and local livestock interests over range burning and free-ranging livestock, especially hogs. While economic interests of lumber companies, foresters, and some absentee landowners focused on timber, the economic interests of the local people were centered on the understory grasses for cattle grazing and, in the pinelands, on turpentine production. As early as 1850, the state geologist of Mississippi reportedly stated "The beautiful park-like slopes of the pine hills are being converted into a smoking desert of pine trunks on whose blackened soil the cattle seek more vainly every year the few scattered sickly blades of grass whose roots the fire has not killed" (Dunston 1913). A U. S. Forest Service examiner (Dunston 1913) reported three-fourths of the woodlands in Mississippi were burned over every year and many localities were burned twice or more. He called for a vigorous education campaign, stating "The enormous loss caused by the boll weevil is not greater than that from forest fires, nor is the extermination of the cotton pest of more vital importance to the future wealth of Mississippi than the prevention of such fires."

Similar conditions and concerns were reported for pinelands elsewhere. W. W. Ashe (1894) made a survey of forest conditions in eastern North Carolina for the North Carolina Geological Survey and reported a chief consideration in the regrowth of longleaf pine in North Carolina to be "entire prevention of all fires among the pines." In another report on forest conditions in North Carolina, Gifford Pinchot and Ashe (1897), stated "The first and absolute prerequisite before any attempt can be made to improve the condition of the long leaf pine forests is entire exclusion of cattle and hogs and complete protection from fire."

The hardwood areas of the mountains were similarly burned and grazed. Regarding forest conditions in the mountain region of North Carolina at the turn of the century, Holmes (1911) reported unrestricted grazing despite local stock control laws in some counties. He reported 20 to 50 percent of the forest land in each county was burned annually "with the false idea that (fires) improve the range."

Throughout much of the South, Coastal Plain and mountains, large lumber companies began extensive commercial harvest of timber in the late 1800's. The wasteful and destructive logging pattern, typical of the time, left behind vast amounts of slash and logging debris. Annual burning of the cutover lands continued, intensified by the heavy fuel loads.

From the foresters' perspective, there was reason to be concerned. Foresters correctly perceived that uninterrupted annual burning did not allow forests to regenerate. However, many of their other ideas proved to be wrong—for example, claims that fire should be totally excluded from longleaf pine forests (Ashe 1894, Pinchot and Ashe 1897), that it resulted in "soil impoverishment" (Dunston 1913), that it resulted in short-bodied, limby, knotty longleaf pine trees of low value (Ashe 1894), and that burning actually harmed the grazing resource rather than improving it as the woods-burners contended (Dunston 1913; Holmes 1911).

In this political environment it is not surprising that Stoddard's articulate advocacy of controlled burning was fiercely opposed, especially by foresters. The U. S. Forest Service maintained a firm policy against burning (chronicled by Schiff 1962). Federal funds available for state forestry agencies under the Clarke-McNary Act of 1924 were withheld from states if they tolerated controlled burning. The American Forestry Association sponsored a massive propaganda campaign, the Southern Forestry Education Project, from 1927 to 1930. Teams of men, known as the "Dixie Crusaders" were sent into the rural South with trucks equipped with generators, movie projectors, films, radio broadcasts, posters, and pamphlets. According to Schiff (1962), "the crusaders traversed 300,000 miles, disseminating 2 million pieces of literature along the way. More than 5,200 motion picture programs and lectures were presented to 3 million people…." Damage to wildlife was emphasized in the campaign.

Also, beginning in the 1930's, the Forest Service employed sociologists and psychologists to study the woods-burners and continued to support research on the subject for more than 40 years. In mostly unpublished reports to the Forest Service (summarized by Pyne 1982), the researchers concluded that underlying reasons and motives for woods burning included social isolation, boredom, ritualistic tradition ("our pappies burned the woods," Shea 1940), frustration of a culturally and economically disadvantaged group, alienation, and creation of jobs in fire suppression.

The developing pulp and paper industry added its influence to the anti-fire effort. In the 1940's the National Advertising Council, the U. S. Forest Service and state forestry agencies created what has been called the most effective advertising campaign in history: the Smoky Bear program. And, in an appeal to the strong religious convictions held by most rural southerners, compilations of Bible passages deemed to be anti-fire were printed in pamphlets and widely distributed (e.g. U. S. Department of Agriculture 1955) and at least one poster showed Smoky Bear praying for people to be careful with fire.

At the time of his original study of bobwhite quail, Stoddard was an employee of the Bureau of Biological Survey in the U. S. Department of Agriculture, and publications by employees had to go through review by other agencies in the Department that might be affected. Stoddard had great difficulty in getting the chapter on controlled burning in his bobwhite quail book past reviewers in the U. S. Forest Service and had to revise it several times (Stoddard 1969).

In his memoirs, Stoddard (1969) wrote that the anti-fire campaign was "the most intensive—and ludicrous—educational campaign that ever insulted the intelligence of American audiences. It was carried on by well-meaning but utterly misinformed persons."
Stoddard wrote:

"One of the main arguments against the custom of burning off the longleaf forests was the alleged disastrous effect on game and wildlife in general. . . Motion pictures of deer with hair and hooves burned off in disastrous crown fires (in the west or the Lake States) were regularly shown on southeastern screens…. Dead fish, killed by the ash from explosive mountain fires, were shown rotting in the sun as examples of what would happen if the Florida cattlemen did not stop firing the flatwoods."

Stoddard complained that, as forestry agencies gradually withdrew their unalterable opposition to the use of fire "in a gesture that might be termed face-saving, they substituted the expression 'prescribed burning' for 'controlled burning,' and insisted that experts must 'prescribe' the practice."

In their own defense, foresters noted that burning was not easily confined within property boundaries, and they contended that advocating prescribed burning while conducting a program to persuade people to reduce the occurrence of wildfire would send a mixed message and weaken fire control programs. Also, they noted that advocates of prescribed burning did not have the responsibility of controlling destructive fires as did foresters. (Reibold 1971).

Acceptance of Prescribed Burning by Public Agencies

U. S. Forest Service policies toward prescribed burning were especially important to wildlife management during the first half of the century because the Forest Service at that time controlled most public land important to wildlife in the South, including most wildlife management areas, which were under cooperative agreements with state wildlife agencies. It also greatly influenced state and private actions through funding programs and cooperative work with state forestry agencies and educational programs.

Reibold (1971), in reviewing the history of prescribed burning from the perspective of the Forest Service, explained that a major reason that forestry agencies were slow to put prescribed burning into practice was inadequate budgets and professional staff, coupled with the need to address other priorities. In the 1930's, when large-scale acquisition of national forests was undertaken with little professional staff and inadequate budgets, priority had to be given to hundreds of transactions involved in land acquisition, mapping, development of work programs for Civilian Conservation Corps camps, reforestation, and fire control. Also, Reibold contended that proper execution of prescribed burning was not possible until tractor plows became generally available, which was not until after World War II.

But Reibold (1971) acknowledged that there was much resistance among foresters to prescribed burning. Nearly all of the early foresters in the South were from the North and the West. A strong German protectionist influence in the forestry schools was greatly reinforced by experiences with the extremely destructive wildfires that followed logging in the northern states, burning millions of acres and, in several cases causing hundreds of human deaths. Reibold said "some (foresters), having spent a lifetime in working to prevent fires or in fighting them, were emotionally opposed to what seemed to them to be an abandonment of all they had worked for."

Early in the Twentieth Century, some scientists and professional foresters had begun to recognize the role of fire in maintaining longleaf pine forests. The contributions of a few individual botanists, foresters and animal husbandrymen, who bucked the tide, are detailed by Harper (1962), Pyne (1982) and Schiff (1962). Although administrators reportedly suppressed and delayed publication of research on controlled burning (Pyne 1982; Schiff 1962), some research personnel in the U. S. Forest Service were reporting on the effects of burning and its possible use in silviculture. A listing and abstracting of publications of the Southern and Southeastern Forest Experiment Stations from 1921 to 1955 (Bruce and Nelson 1957) shows publications dealing with the benefits of prescribed burning in forage production, pine regeneration, fire hazard reduction, control of brown spot needle rust, and hardwood control appearing as early as 1932 and becoming numerous in the 1940's. There were proposals and directives for limited prescribed burning on federal lands in the 1930's (Reibold 1971). But, with a few exceptions, research findings and policy directives were not implemented, and prescribed burning for the most part was restricted to private lands. There was little use on public lands until the 1960's, and even the forest industries were reluctant to do much prescribed burning.

It was not until after World War II, when equipment and manpower became available that the national forests began to undertake burning, mostly for fuel reduction. In 1947 prescribed burning was initiated on the Francis

Marion National Forest, including some growing season fires for hardwood control and periodic winter fires to improve habitat for wild turkeys (*Meleagris gallopavo*) (Devet and Hopkins 1967; Holbrook 1952). This was one of the earliest cases of prescribed burning on a national forest with wildlife habitat improvement as an objective. By the decade of the 1950's, prescribed burning on the national forests averaged about 250,000 acres per year (Reibold 1971). During the 1960's, the U. S. Forest Service cautiously began expanding prescribed burning on forests outside the longleaf pine region.

In 1971 the U. S. Forest Service sponsored a symposium on prescribed burning. In the foreword to the proceedings (USDA 1971) symposium chairman Stephen Boyce stated that among the 450 scientists, land managers, and environmentalists attending, "consensus was essentially unanimous that prescribed fire, when properly used in the South, is an almost indispensable management device having generally beneficial effects, certainly lacking in sustained deleterious effects on the crop trees, on the soils on which they grow, or on the flora and fauna of the area burned." Symposium participants, however, did recognize a lack of knowledge about the effects of prescribed burning on air quality—a matter of growing concern.

Prescribed burning had not been as big an issue with other federal agencies because they controlled few areas with upland forests in the South. There were only four national wildlife refuges with large areas of upland forest. There was some prescribed burning on the refuges in the 1940's, but generally for uplands there was a fire exclusion policy from 1949 until even-aged forest management was adopted in 1962 (Czuhai 1981; Givens 1962). Prescribed burning in waterfowl marshes (Lynch 1941) had been accepted much more readily than on forestland (Givens 1962).

The National Park Service had a strong anti-fire policy from the time of its establishment in 1916 until 1958 when a policy change was made specifically for the Everglades National Park and the first prescribed burn in a national park was conducted that year. The antifire policy of the National Park Service was reversed in 1967 (Hendrickson 1972; Kilgore 1974; Taylor 1981) to allow prescribed burning where necessary for community restoration and maintenance.

At the state level, most wildlife management areas were not owned by the states but were operated under cooperative agreements with landowners, who approved or disapproved habitat management measures. Before about 1960, most management areas in the Southeast were on national forests, many of them in the mountains. Thus, state wildlife agencies had little opportunity to carry on prescribed burning.

Attitudes Toward Prescribed Burning Within the Wildlife Management Profession

Primarily because of Stoddard's influence, backed by his rigorous studies of bobwhite quail, wildlife biologists in the South had accepted prescribed burning, at least in principle, before other resource management professionals. The basic techniques of controlled burning for game management were well established by the mid-1930's. The optimum season, frequency, and conditions for burning for various game species and important food plants were recognized. And special techniques, such as spot-burning at night to create a mosaic of vegetation, were in use (Stoddard 1935b). Prescribed burning on an annual or biennial rotation was a standard practice in the management of bobwhite quail. In 1932 Stoddard reported "occasional controlled burning of overly 'rough' cover" was among "measures practiced on scores of Southeastern preserves. . ." for wild turkeys (Stoddard 1932); he recommended specific burning techniques on a 2- to 4-year rotation (Stoddard 1935a; 1939). And, although restoration of white-tailed deer (*Odocoileus virginianus*) populations in the South was just beginning, controlled burning was used to improve browse conditions in areas of the Coastal Plain. As previously noted, burning of marshes to improve habitat for waterfowl and muskrats (*Ondatra zibethica*) was in practice in the 1930's or earlier (Lynch 1941).

Outside the South, wildlife biologists were skeptical or more cautious. Despite research showing improved habitat conditions for many species following experimental and wild fires, prescribed burning was not commonly employed for the management of wildlife. With few exceptions, books on wildlife management before 1960 treated prescribed burning as a practice peculiar to the South. From 1930 through the 1950's there were seven important textbooks on wildlife management. With one exception, all of them emphasized protection from fire and treated prescribed burning as a practice limited in application to the southern coastal plain.

In his classic textbook "Game Management," Aldo Leopold (1933), who was well acquainted with Stoddard's work, made only passing mention, in a scattered half dozen sentences, of controlled burning. He referred to spot burning of heather for red grouse (*Lagopus lagopus*) and to Stoddard's use of fire in bobwhite quail management to control nesting cover. He mentioned Stoddard's suggestion that controlled burning may reduce ectoparasites and might prove useful as a means of range sanitation, but then wrote "Use of fire without sound evidence of its effects, or on lands primarily devoted to other crops, is of course to be deplored."

In "The Land and Wildlife" Edward H. Graham (1947) briefly acknowledged the use of fire in management of

Gulf Coast marshes for muskrats and snow geese (*Chen caerulescens*). Regarding use of prescribed burning on forest lands, he wrote only two sentences: "It has been used in longleaf pine forests of Georgia to check growth of hardwoods and increase wild legumes, a device for improving habitat for bobwhite quail" and "although fire is a tool sometimes employed by both foresters and wildlife managers, its use is specialized and does not negate the general rule of protection from burning."

Reuben Trippensee (1948), in his widely used textbook, referred to controlled burning as "a management technique peculiar to the Southeast, where certain vegetative cover types not only permit the use of fire in this manner but apparently require it if quail range is to be kept productive." Citing Stoddard (1931, 1939), he devoted nearly two pages to controlled burning, but only for quail management in longleaf pine. Regarding wild turkeys, he stated "in all regions except the Southern Coastal Plain, lack of forest-fire control means exclusion of the turkey."

Wallace Grange's book "The Way to Game Abundance" (1949) provided the one notable exception in its treatment of fire as a habitat management tool. Grange considered animal cycles in the North to be largely a result of plant succession linked to a cycle of drought and associated fires. An entire chapter was devoted to controlled burning for accomplishing a variety of objectives in the management of various species. Grange stated "with the possible exception of the desert, all other North American game habitats have been, from time to time, improved by the agency of naturally occurring fire, and may be benefited through the application of controlled fire."

Leonard Wing's textbook "Practice of Wildlife Conservation"(1951) dealt similarly with the subject. Wing made the statement that "The value of controlled burning is becoming recognized, but a clear distinction should be made between *controlled* and *uncontrolled* burning." He then devoted about a page to the practice of controlled burning in the Southeast. He gave no examples from other areas but did suggest that controlled burning might be useful in the management of some other game birds and specifically mentioned songbird and non-game conservation.

As late as 1959 Ira Gabrielson, first director of the U.S. Fish and Wildlife Service and long-time president of the Wildlife Management Institute, in his textbook "Wildlife Conservation" mentioned prescribed burning in a paragraph devoted to Stoddard's work with quail, then added "Generally speaking, however, on the basis of present knowledge, fire is so great an enemy of both wildlife and forests that there is an increasing amount of fire control, both on public and on private lands. This should in the long run be exceedingly beneficial to wildlife" (Gabrielson 1959).

Into the 1980's most textbooks and references on wildlife management continued to reflect little awareness of or interest in prescribed burning.

Roy Komarek (1966) criticized wildlife professionals for neglecting habitat management in general and prescribed burning in particular. He noted that the index to the first 29 volumes of the Journal of Wildlife Management had no entries for "controlled burning" or "prescribed burning", and only seven entries to "burning" or "fire." He found the subject similarly lacking in the other publications of the Wildlife Society. Writing that the wildlife profession was lagging in the use of fire, he called for more experimentation with season, frequency, and size of prescribed fires for different objectives and different species, including non-game species and wild flowers. Previously a strong critic of the conservative attitude of foresters toward controlled burning, he noted that by this time foresters had perfected techniques by which they were burning hundreds of thousands of acres. He wrote "Despite the fact that Stoddard pioneered the use of controlled burning on game lands more than thirty years ago, the field of wildlife management, except in a few cases, has contributed little to the advancement of the art and in many cases depends upon techniques developed by the foresters."

Continuing Stoddard's promotion of the study of fire ecology and the controlled use of fire in land management, the Tall Timbers Research Station was formed in 1958, and through its research, demonstration plots, and especially its annual fire ecology conferences and published proceedings, contributed greatly to the study, understanding and use of fire. Stoddard, brothers E. V. and Roy Komarek, and others at Tall Timbers deserve much credit for the acceptance of prescribed burning outside the Deep South. At the first North American Wildlife Conference, Stoddard (1936) had urged experimentation with prescribed burning in other regions. And, from the beginning, the fire conferences were worldwide in scope, bringing in speakers from all continents to address the ecology and use of fire in many vegetation types and parts of the world, emphasizing that the usefulness of prescribed burning was not limited to management of longleaf pine.

Prescribed Burning in Nongame Wildlife Management

Although management of nongame wildlife is a relatively new emphasis, recognition of the value of fire as a potential tool for nongame wildlife management is not. Most prescribed burning has multiple objectives, and just because prescribed burning was not conducted specifically for nongame objectives does not necessarily mean that managers were unaware of the effects on nongame species and did not take them into consideration when burning. In his early writings, Stoddard, who was an esteemed ornithologist, made frequent references to effects of fire on species other than game, forage, and commercially important timber. In one of his early publications (Stoddard 1936) he wrote that "multitudes of small birds" use the burns in late winter and early spring and "such burns (are) important to their welfare also." In one of his reports to cooperators, he included a section titled "burning to maintain floral beauty of pinelands" (Stoddard 1939). Stoddard's recommendations for burning in quail management included discussion of fire's effects on rodent and predator populations and its role in reducing predation on quail. E. V. Komarek, working with Stoddard, elaborated on the effects of controlled burning on mammals in a paper for the Journal of Mammalogy (Komarek 1939). Komarek's later writings included frequent references to the benefits of fire to wildflowers, earthworms, butterflies, and birds (Komarek 1969, 1971).

But, until the 1970's there was little emphasis in the South or elsewhere in managing specifically for nongame wildlife (Landers and Johnson 1980), so nongame species were only rarely a specific objective of prescribed burning. Interest in managing habitats for species other than game animals greatly increased in the 1970's. This was evident in the surge in conferences, symposia and special sessions at technical meetings devoted to nongame species (DeGraaf 1978; Odom and Guthrie 1981; Odom and Landers 1978; Thompson 1971). Although most effort in nongame and endangered species management at that time was devoted to determining the status of these species, prescribed burning was considered in papers making recommendations for habitat management for various reptiles and amphibians (Herman 1981; Landers and Speake 1980; Means and Campbell 1981; Means and Moler 1978), nongame birds (Edwards 1978; Meyers and Johnson 1978; several papers in Thompson 1971; Wood and Niles 1978) and mammals (Hilliard 1979; Lustig and Flyger 1975; Winchester et al. 1978).

Also, it was recognized that prescribed burning for forestry or game management objectives maintained a biotic community with a distinct suite of plant and animal species (Komarek 1971; Landers and Speake 1980), and prescribed burning received increasing attention for community restoration and maintenance in parks and natural areas (Hendrickson 1972).

We end this historical review with a symposium on prescribed fire and wildlife in southern forests held at Myrtle Beach, South Carolina in 1981 (Wood 1981). The symposium included papers on the status of prescribed burning programs on state, federal, and private lands and a series of papers reviewing the database on the effects of prescribed fire on many individual species or groups of vertebrates. The presentations demonstrated that, although there was generally strong conceptual understanding of fire ecology and its relation to habitat needs of various wildlife and there was a good database for some game species, with a few exceptions, data on fire and non-game species were seriously inadequate.

Implications

The most obvious lesson of the controlled burning controversy in the first half of the Twentieth Century should be a reminder that all knowledge does not reside with professionals or government agencies. In 1981 E. V. Komarek noted that private hunting preserves of the Southeast for more than 50 years had burned between 750,000 and 900,000 acres annually, leading the way in spite of intense discouragement by government agencies and many resource management professionals (Komarek 1981).

Secondly, this review shows that conditions are always changing. Fire has been a significant environmental factor throughout much of North America for thousands of years. But its role has been a dynamic one—determined by climatic shifts and for several thousand years by the ever-changing human factor. There is much interest now in managing for natural landscapes. But, it is not possible to define what was *the* natural condition, except by defining the exact place and time—and accepting human influences as natural. Managers could set a goal of re-creating conditions of the sixteenth century when the Spanish explorers passed through the South. But, early descriptions of the landscape are spotty and probably biased; and, even if early conditions were known, they might not be what is desired today. Early accounts indicate that Indians used fire excessively in many areas, and the landscape they created in some areas may have excluded forest cover that would be valued today. Furthermore, in most places it would not be possible to re-create prehistoric conditions because soils have been drastically changed, important species have been lost, and exotics have become established. Therefore, we think prescribed burning strategies should be based on objectives that are better defined than vague notions of what is natural.

Third, land managers must be able to adapt to continuing change. With increasing human populations and urban-suburban encroachment on rural areas, there will be increasing concerns about public health, automobile accidents, and liability resulting from problems with air quality and smoke produced by prescribed burning. These concerns may result in serious limitations on the use of prescribed burning in the future. And, if, as many scientists contend, the climate is becoming warmer and drier, this may drastically alter the use of fire—for example, making hazard reduction a more important objective and more seriously restricting the season of burning. Also, history shows that priorities in resource management are controlled by economic and political conditions. Controlled burning conducted by herdsmen 100 years ago was well suited to their objective; but with different economic conditions (i.e., development of the forest industry), changes were needed. Today, much prescribed burning has multiple objectives, even in wildlife management. Current interest in management for game and nongame wildlife and natural areas may wane if economic prosperity declines or there is an urgent need for other forest products. Therefore, development of techniques for a diversity of approaches to fire management is needed to allow compromise and adaptability in meeting future needs. Research also must continue on alternative methods of vegetation control because the use of prescribed burning will almost certainly become restricted in some areas.

Finally, fire research is still needed. Managers have been able to draw on 300 years of burning experience and more than 50 years of fire research in the South. But, since the 1981 symposium on prescribed fire and wildlife, prescribed burning has been applied to an increasing variety of situations where new objectives are in place. Management objectives include an increasing variety of plant and animal species and communities. Prescribed burning is no longer considered to be out of bounds of consideration in the management of any habitat type. It is being used, at least in a limited way, in the mountains for site preparation, hazard reduction, oak regeneration, maintenance of bog turtle (*Clemmys muhlenbergi*) habitat, maintenance of grassy balds as habitat for small mammals and raptors, and management of plants of special interest. Growing season fires are being used increasingly for hardwood control, promotion of flowering of grasses and other forbs, and to produce diversified patterns of plant growth. Infrequent high intensity fires are being used to restore or regenerate communities, and new technology for igniting and controlling fires is being developed by foresters. With more diverse objectives and increased use of prescribed burning in habitats other than southern pine forests, the need for research is greater than ever, and the research database may be weaker in relation to needs than it was in 1981, when the prescribed fire and wildlife conference was held. Roy Komarek's criticisms of the wildlife profession in 1966 remain valid today. We think this is especially true outside the South.

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Fire, Herbicide, and Chainsaw Felling Effects on Arthropods in Fire-Suppressed Longleaf Pine Sandhills at Eglin Air Force Base, Florida

Louis Provencher[1], *Krista E. M. Galley*[1], *Andrea R. Litt*[1], *Doria R. Gordon*[2],
Leonard A. Brennan[3], *George W. Tanner*[4], *and Jeffrey L. Hardesty*[2]

Abstract.—Experimentally evaluating the success of hardwood reduction techniques against a "model" reference condition of longleaf pine sandhill communities is not directly possible because reference sites are not randomized or replicated. We addressed this issue by measuring the similarity of arthropods in treatment (fire, herbicide, felling/girdling, and control) and reference sites using three indices. Arthropod assemblages from plots that burned were significantly more similar to those in the reference condition than were those that did not receive fire. Our findings suggest that increasing arthropod prey using fire will be beneficial both to management and to biodiversity.

Introduction

Southern longleaf pine (*Pinus palustris* P. Mill.) forest restoration is, in many ways, essentially synonymous with red-cockaded woodpecker (*Picoides borealis* Vieillot) habitat management, where mechanical removal and short burn rotations (1–3 years) are used to reduce and control midstory hardwood encroachment that is detrimental to the woodpeckers (Conner and Rudolph 1989; Hooper et al. 1991). Because many public lands in the southern U.S. are now managed for red-cockaded woodpeckers (Brennan et al. 1995; U.S. Forest Service 1995), management effects on non-target species, especially birds, have become a concern in recent years (Brennan et al. 1995; Burger et al. 1998; Hunter et al. 1994; Plentovich et al. 1998; White et al. 1999).

Red-cockaded woodpecker habitat management (i.e., fire) likely affects arthropods directly or indirectly by stimulating population growth through increased resource availability and quality (James et al. 1997; Reed 1997). It is well documented that fire causes plants to resprout and to be more palatable to herbivores (smaller C:N ratio for plant tissues) (Dunwiddie 1991; Nagel 1973; Owensby et al. 1970; Smith and Young 1959; Stein et al. 1992). Increased flowering after fire (Platt et al. 1988) should attract insect pollinators. In turn, increases in herbivores and pollinators may attract predatory arthropods and parasites. Some predatory ground beetles, for example, increase following fire (Harris and Whitcomb 1974). Increases in arthropods are important because they represent a fundamental prey base for red-cockaded woodpeckers and associated vertebrates, many of which exclusively feed arthropods to their offspring and depend on arthropods for food outside of the breeding season. Conversely, these species of concern may suffer if fire suppression results in decreases in arthropod abundance. Remarkably, there is a dearth of quantitative data published on the relationship between fire and the arthropods that red-cockaded woodpeckers consume (but see Hanula and Franzreb 1998; James et al. 1997) or on arthropods in longleaf pine forests (reviewed in Folkerts et al. 1993), especially describing old-growth communities.

We experimentally compared the effects of three hardwood reduction techniques (growing season burn, herbicide application, and midstory mechanical felling/girdling) and a no-treatment control on herb-layer arthropod densities in fire-suppressed longleaf pine sandhills at Eglin Air Force Base (EAFB), Florida. Hardwood reduction is necessary to meet EAFB's goals for restoring functional, diverse sandhill systems across the Base and for habitat restoration for the endangered red-cockaded woodpecker and other target species having similar habitat requirements. Our main goal was to identify which management technique(s) caused arthropod species assemblages to converge toward values found in fire-maintained sandhills, which we termed the "reference condition" to emphasize forest composition and function in addition to structure. Based on our analysis, we also developed metrics that may be indicators of sandhill management success.

Methods

Site Description

Eglin Air Force Base (EAFB) occupies 187,555 ha in the southern portions of Santa Rosa, Okaloosa, and Walton counties in the western Florida Panhandle. The mean annual temperature is 18.3° C, with approximately 275 freeze-free days per year. Mean annual precipitation is 158 cm. Monthly precipitation levels peak slightly during late spring and early summer months and decrease during the winter months (Chen and Gerber 1990). Since 1886, 25 tropical storms and 27 hurricanes made landfall within 97 km (60 miles) of EAFB.

The terrain is level to gently rolling with occasional areas of steep slopes along creeks. Elevation ranges from 0–100 m above sea level, and the landscape generally slopes to the southwest toward the Gulf of Mexico. Lakeland series, the common soil, is a medium to fine sand with 5-10 percent silt and clay (Overing et al.

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[1] The Nature Conservancy, Niceville, FL 32588
[2] The Nature Conservancy, Gainesville, FL 32611
[3] Tall Timbers Research Station, Tallahassee, FL 32312
[4] Department of Wildlife Ecology and Conservation, University of Florida, Gainesville, FL 32611

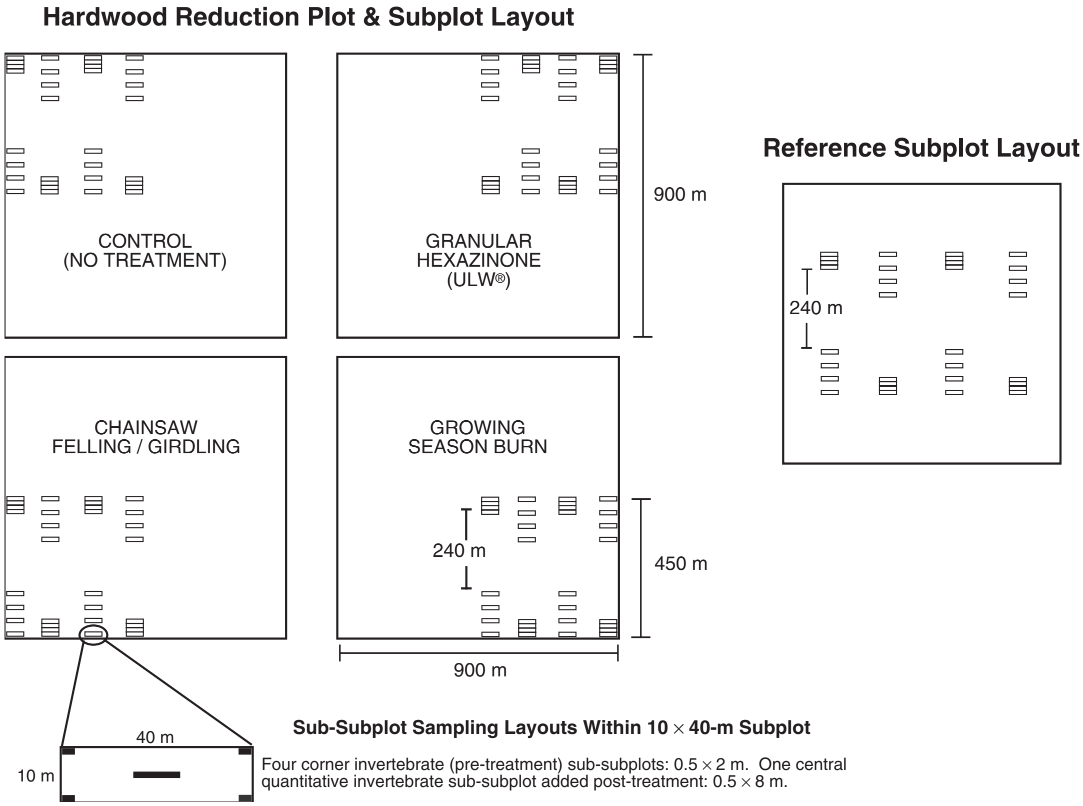

Figure 1.—Sample layout of one of six reference plots (81-ha) and of hardwood reduction plots and sampling areas in one of six blocks in a randomized complete block split-plot design consisting of four whole-plot treatments. Spatial randomization of treatments varies per block.

1995), is rapidly permeable and strongly acidic, with nearly level to steep slopes. The vegetation of EAFB's sandhills is described in Rodgers and Provencher (1999).

Restoration Experiment: Experimental Hardwood Removal Plots

A total of 24, 81-ha plots were established in six blocks of four fire-suppressed hardwood-longleaf pine sandhill plots across EAFB (see map in Rodgers and Provencher 1999). This study utilized a randomized complete block design (Steel and Torrie 1980), where each plot within an experimental block was randomly assigned to either control designation (no treatment), or to one of three restoration treatments applied during the spring and early summer of 1995: growing season burn in May or June, herbicide application (ULW®, the granular form of hexazinone with 75 percent active ingredient applied at 2.24 kg/ha), and oaks and sand pine felling/girdling by chainsaw (slash not removed). Fuel reduction burns were conducted in the herbicide and felling/girdling plots in early 1997. In each restoration plot, 32, 10 ´ 40-m subplots were located in the 20-ha corner farthest from the neighboring plots of the block to reduce the potential for recording organisms that can travel across adjacent plot boundaries (Figure 1).

Reference Plots

Three pairs of 81-ha frequently burned, longleaf pine-dominated reference sandhill plots were established (Fig. 1). These plots were not part of the experiment described above but were used to measure restoration success. In each reference plot, 32, 10 ´ 40-m subplots were located in the center of the plot to reduce edge effects (Figure 1). These plots are more fully described in Rodgers and Provencher (1999) and Provencher et al. (2000; 2001a; 2001b). For the duration of the restoration study, reference plots were under a "let burn" management policy (all reference plots burned once and four plots burned at least twice during the study period).

Data Collection

Densities of selected herb-layer arthropod species were estimated. To successfully collect arthropods of various sizes and mobility, individuals were first collected using a sweep net, followed immediately by a modified D-Vac insect vacuum. In 1994, individuals were collected from herb-strata vegetation (<1.4 m) within four 0.5 ´ 2-m sub-subplots (Figure 1). Because we suspected that the noise and motion of our vacuum device and sweep net were flushing some arthropods from adjacent sub-subplots, we changed the location and shape of arthropod sampling sub-subplots to a single 0.5 ´ 8-m rectangle in the center of the subplot beginning in fall 1995 (Figure 1). Moreover, the second method minimized escapes by arthropods because we opened the sweep net once instead of four times. Specimens from sweep net/D-Vac samples were manually sorted and preserved in 70 percent ethanol. We enlisted the assistance of taxonomic specialists, where possible, to perform initial species identifications and established a reference collection of more than 300 species of authoritatively identified adult arthropods.

Statistical Analyses

Similarity measures allowed us to directly determine which treatment plots most resembled the reference plots. Treatment differences were tested using two-way analysis of covariance (ANCOVA). The advantages of this method over discriminant function analysis were that (1) we could include both common variables and those with patchy distributions and (2) the calculated similarity values, rather than the raw data, were required to meet any statistical assumptions of the tests. Although there are a wide variety of available similarity indices, we calculated similarity of selected variables between each treatment plot and each reference site using three formulas (Brower et al. 1989; Underwood and Chapman 1998). The purpose of using three indices was to achieve a consensus among them, especially for the identification of indicator variables (see below). The strength of an indicator increased with the number of indices identifying it as a significant contributor to the pattern of similarity.

Proportional similarity (PS) was selected because it is widely used and it accounts for the relative abundance of variables. This last feature was especially important to us because site conditions (e.g., soil productivity) may greatly change the values of certain variables but not their relative abundance on a particular sampling site. The PS between each treatment plot i (= 1,…, 24) and each reference plot j (= 1,…,6) was averaged over all reference plots per restoration plot i with a sum weighted by sample sizes;

$$PS_{ij} = \sum_{j=1}^{6} n_j \, (1-0.5\sum_{k=1}^{K} |p_{ik}-p_{jk}|)/N$$

where p_{ik} is the proportion of the logarithm of variable k in treatment plot i, p_{jk} is the proportion of the logarithm of variable k in reference plot j, n_j is the number of subplots in reference plot j, and N is the total number of subplots in all reference plots (Brower et al. 1989). We took the logarithm of variables to prevent large values from dominating PS and, thus, to increase the representation of uncommon species. Plots that share all the same variables in the same proportions will have a PS = 1, whereas plots that share no variables will have a PS = 0.

Following Underwood and Chapman's (1998) description of its superior properties, we calculated the weighted average of the 1 – Bray-Curtis index of dissimilarity,

$$BC_i = \sum_{j=1}^{6} n_j(1-\sum_{k=1}^{K} |Z_{ik}-Z_{jk}|/(\sum_{k=1}^{N} Z_{ik} + \sum_{k=1}^{N} Z_{jk}))/N$$

where Z_{ik} is the mean abundance of species k in plot i. This index does not require proportional abundance or any transformation and is, therefore, more intuitive than proportional similarity, but it is interpreted in the same way as proportional similarity.

We created a third similarity index, endpoint difference (ED), bounded by 0 and 1 to incorporate within-plot variation,

$$ED_i = \sum_{j=1}^{6} n_j \sum_{k=1}^{K} \exp[-|Z_{ik}-Z_{jk}|/\sigma_{eij}]/K/N$$

where the value within the exponential function is the absolute value of the *t* statistic (Steel and Torrie 1980), s_{eij} is the joint standard error of Z_{ik} and Z_{jk} assuming unequal sample sizes, K is the number of variables, and Z_{ik} is either the average of a single variable (no logarithmic transformation) k from plot i (similarly for plot j) or the average proportional value of variable p_{ik}. of variable k. Plots that share the same variables in the same proportions (all $|Z_{ik}-Z_{jk}| = 0$) or that have large variability (standard errors are large) preventing these plots from being differentiated will have an ED equal or close to 1. Plots that share no variables (all $|Z_{ik}-Z_{jk}|$ are large) or that can be easily distinguished due to low variability (standard errors approach 0) will have an ED close to 0.

To determine which variables contributed most to the similarity pattern among treatments, we correlated the 24 similarity values with the contribution of each species. A positive correlation would indicate that the variable supported the similarity pattern, whereas a negative correlation would mean that the variable weakened the similarity pattern. We retained only variables with significant correlations (≥ 0.481 or ≤ -0.482, df = 23, $P < 0.05$) (Steel and Torrie 1980). For proportional similarity, the contribution of each variable per plot *i* was arbitrarily measured by:

$$\sum_{j=1}^{6} (1-0.5|p_{ik}-p_{jk}|)/6$$

where we averaged contributions over the six reference sites per variable. Similar averaging and calculations were performed for the 1 - Bray-Curtis and endpoint difference contributions:

$$\sum_{j=1}^{6} \left(1-|Z_{ik}-Z_{jk}|/\left(\sum_{k=1}^{N} Z_{ik}+\sum_{k=1}^{N} Z_{jk}\right)\right)/6, \text{ and}$$

$$\sum_{j=1}^{6} \exp[-|Z_{ik}-Z_{jk}|/\sigma_{eij}]/6, \text{ respectively.}$$

We tested restoration treatment effectiveness by comparing the similarity indices using two-way ANCOVA for a randomized complete block design (Steel and Torrie 1980). Pre-treatment similarity was used as the covariate to account for differences among plots that existed prior to treatment application. Keeping within the maximum number of allowable independent contrasts for three degrees of freedom (Sokal and Rohlf 1981), we contrasted the following treatments: control versus spring burn (C vs. B), burn versus the herbicide ULW® (B vs. H), and herbicide versus felling/girdling (H vs. F) (see Provencher et al. 2000b for more information). Similarity values were transformed with $\sqrt{x}$, x^2, or $\log(x)$ to stabilize variances when necessary.

Results and Discussion

Similarity to the Reference Condition

Significant increases in similarity for herb-layer arthropods between spring burn and reference plots were not observed until fall 1996, although endpoint difference, while congruent with the other indices, was not significantly affected by treatments (Fig. 2A). The fact that arthropod similarity was only responding to fire was instructive. The arthropods that we sampled appeared to be closely tied to the plants as resources, but not to the percentage of hardwood reduction. In 1996, one year after initial treatment, felling/girdling topkilled the most oaks (93 percent), followed by herbicide (69 percent), and fire (18 percent), compared to fire-suppressed control plots, where the highest oak densities were found (1330 stems/ha ± 60) (Provencher et al. 2001b). On the other hand, fire causes groundcover plants to resprout, providing tender, nutritious forage promoting the growth of plant-eating populations (e.g., grasshoppers, leafhoppers, planthoppers) or groups that feed on live and dead vegetation (e.g., springtails) (Dunwiddie 1991; Nagel 1973; Owensby et al. 1970; Smith and Young 1959; Stein et al. 1992). Herbivorous arthropods and springtails are among the numerically dominant species we sampled, therefore strongly affecting similarity. In turn, a greater availability of arthropods lower in the food web should attract predators and parasites such as spiders, wasps, and some flies and beetles. Indeed, the parasitic (braconid) wasps *Chelonus* sp. and *Heterospilus* spp. and the spiders *Tmarus rubromaculatus* (Keyserling), *Misumenops* spp., *Mimetus* sp., and *Hentzia palmarum* (Hentz) were among the 20 more common species in burn and reference plots during the fall 1996.

Fuel reduction burns applied to herbicide and felling/girdling plots during the winter and spring of 1997 presented several opportunities to test if the response of herb-layer arthropods in herbicide and felling/girdling plots would follow the same pattern observed in the spring burn plots. We suspected that the effect would be stronger as plant resprouting was expected to be more complete (due to the continuous fuel matrix created by oak leaf drop and felled oaks). Burning should result in approximately 50 percent topkill and abundant resprouting (Glitzenstein et al. 1995), while herbicide application should practically eliminate oaks and significantly reduce resprouting.

Spring 1997 was a rather unusual field season as it immediately followed fuel reduction burns. As a result, similarity analyses were potentially influenced by half of the plots being charred. This does not offer a compelling test of the hypothesis. Not surprisingly, the spring burn plots were closest to the reference condition for all indices, but only significantly so for proportional similarity and 1 – Bray-Curtis (Figure 2B).

Arthropod data collected in Fall 1997 showed the strongest convergence to that in the reference condition. Despite the hot fires in felling/girdling-fire plots, the herb-layer arthropod assemblage was more similar to the reference assemblages in the fire only and felling/girdling-fire than in the herbicide-fire plots for all indices (Figure 2B). We attribute the difference between the herbicide and felling/girdling treatments to the greater herbaceous (mostly grasses) cover in the former treatment, a resource that herbivores are expected to exploit. The fire-suppressed control plots were still the least similar to the reference plots. The hypothesis that arthropods would rapidly benefit from the more thorough coverage of the hotter fires was confirmed: arthropods responded within 6 months after the 1997 fires, whereas a response came 1 full year after the spring burns of 1995. Seasonality of fire should not explain this effect because most fuel reduction burns were completed in the early spring. As in 1996, the reduction of hardwoods was not a factor that explained similarity results, the greatest total cumulative topkill of oaks occurred in the herbicide-fire plots (94 percent), with lower mortality in the felling/girdling-fire plots (62 percent) and the prescribed fire only plots (41 percent, a delayed mortality relative to 1996) (Provencher et al. 2001).

Although we hypothesized that the growth of arthropod populations benefited from newly resprouting palatable vegetation, by 1997 this explanation no longer applied to the plots burned two years earlier. This implies that once established, arthropod populations persist beyond when plant palatability is a key factor and/or the continued resprouting of hardwoods (Provencher et al. 2001a) is utilized as a source of palatable vegetation. Our data do not allow evaluation of the duration of fire suppression necessary before insect populations again become depressed.

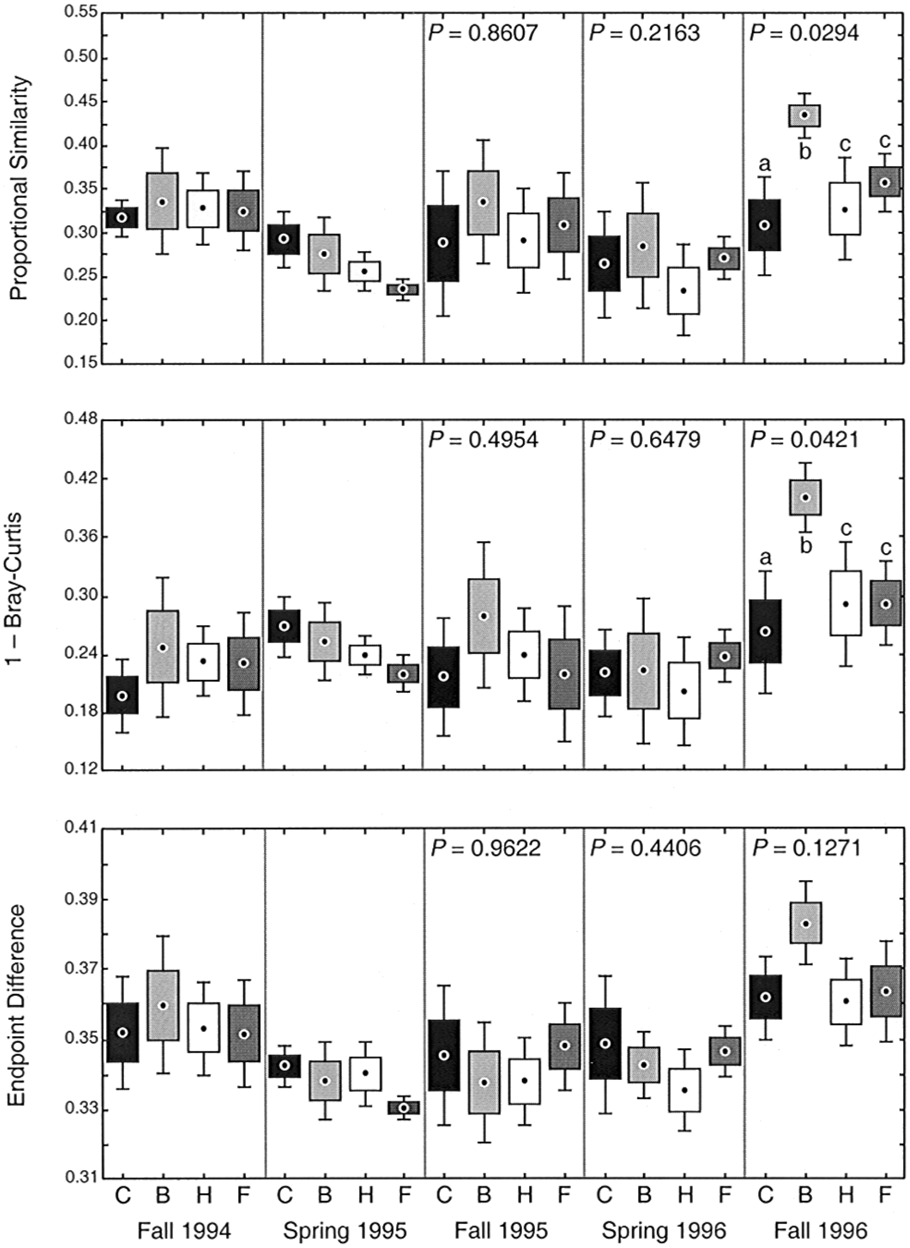

Figure 2A.—Time series of three indices (proportional similarity, 1 – Bray-Curtis index, and endpoint difference) measuring the similarity of herb-layer arthropods between each of four hardwood reduction treatments (spring burn, herbicide, felling/girdling, and no-treatment control) and reference plots from fall 1994-1996 at Eglin Air Force Base, Florida. Tests of treatment effects were calculated with two-way ANCOVA. The experimental design is a randomized complete block (6 blocks), split-plot design, but only the block design at the whole plot level is presented here. The covariate was the pre-treatment data from the fall 1994 or spring 1995. The error term is the mean square of the interaction of the block and restoration treatment effects. Similarity values received various transformations when it was necessary to stabilize variances. The center of the box is the mean, the edges of the box correspond to one standard error, and the error bars are a 95 percent confidence interval.

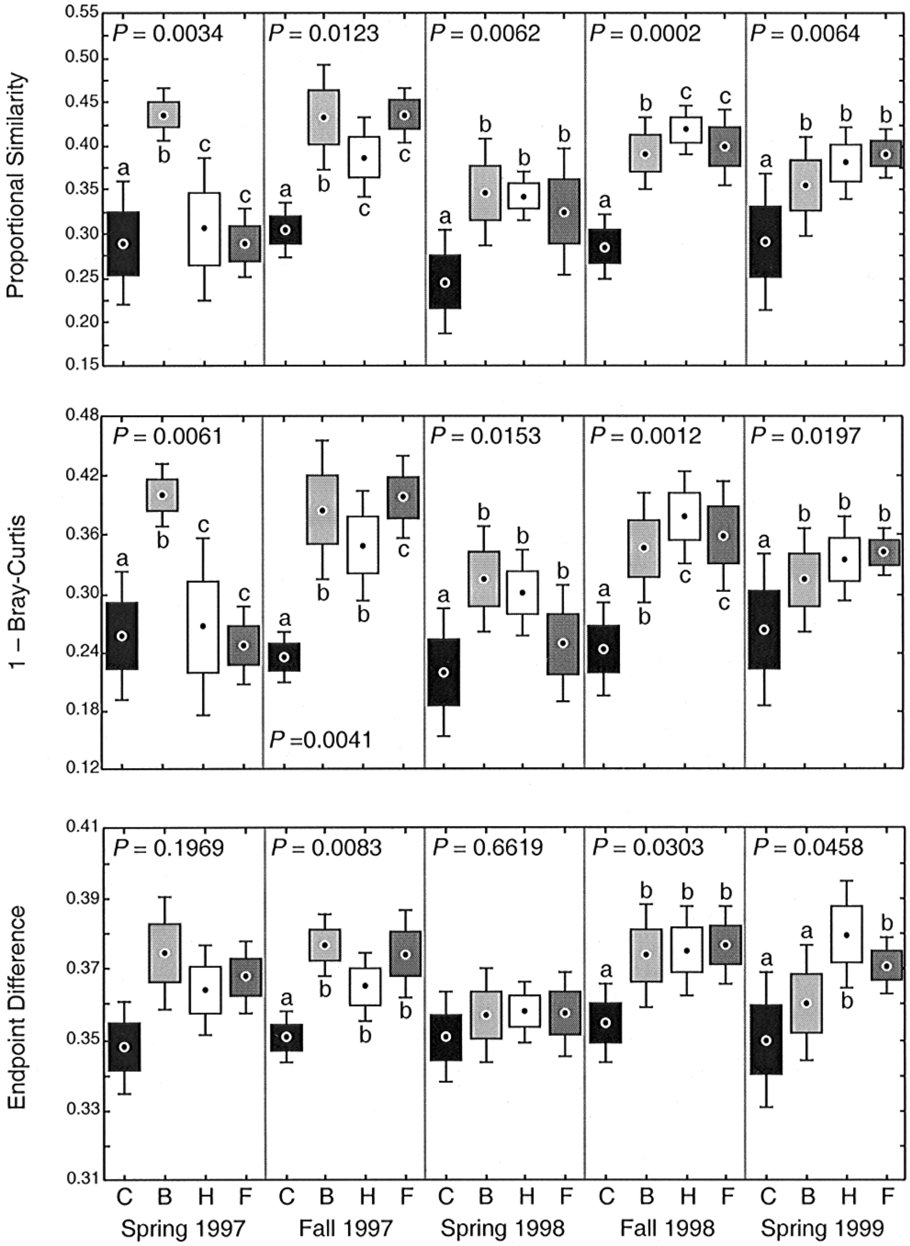

Figure 2B.—Time series of three indices (proportional similarity, 1 – Bray-Curtis index, and endpoint difference) measuring the similarity of herb-layer arthropods between each of four hardwood reduction treatments (spring burn, herbicide, felling/girdling, and no-treatment control) and reference plots from spring 1997-1999 at Eglin Air Force Base, Florida. Tests of treatment effects were calculated with two-way ANCOVA. The experimental design is a randomized complete block (6 blocks), split-plot design, but only the block design at the whole plot level is presented here. The covariate was the pre-treatment data from the fall 1994 or spring 1995. The error term is the mean square of the interaction of the block and restoration treatment effects. Similarity values received various transformations when it was necessary to stabilize variances. The center of the box is the mean, the edges of the box correspond to one standard error, and the error bars are a 95 percent confidence interval.

The similarity patterns seen in 1997 were repeated in spring and fall 1998 and spring 1999, with increased convergence toward the reference condition to some limited extent for herb-layer arthropods in herbicide plots (Figure 2B). Felling/girdling plots showed the same response to a lesser extent (Figure 2B). Fire appears to be the cause of these changes.

Indicators of restoration success

Two valuable outcomes of identifying the species or groups that contributed most to the similarly evaluation (indicator) are to reveal ecological relationships and to identify candidate variables that may be included in a monitoring program. We include as candidate indicators those variables that significantly contributed to the similarity of at least two of the three similarity indices.

Selection of arthropod indicators was difficult because of the great difference among similarity indices. For example, we never identified more than three species with the Bray-Curtis-based index, whereas >10 species were identified by the other indices. The springtail *Sminthurus* sp. 1, the leafhoppers *Erythroneura* spp., the planthopper *Metcalfa pruinosa* (Say), the leafhopper *Jikradia olitoria* (Say), and the jumping spider *H. palmarum* were the strongest indicators and the dominant species to increase after spring burning or immediately after the fuel reduction burns (Fig. 3). Except for the spider, these arthropods are herbivores or detritivores (the diet of *Sminthurus* sp. 1 is unknown but herbivory is suspected). Earlier we explained the relationship between fire, palatable plant resprouts, and arthropod population growth. These indicators illustrate this scenario well. These spiders are generalist predators and may themselves be tracking arthropod prey densities. (Although not evident in Figure 3 for fall 1996, average *H. palmarum* densities were higher in the burn plots than other treatments only after logarithmic transformation.)

During fall 1997, the first time the three similarity measures showed significant treatment differences, *Sminthurus* sp. 1 and the leafhoppers *Empoasca* spp., another common herbivore, were the strongest contributors to all the indices. The jumping spiders *Habronattus* spp. were also positively and significantly correlated to the similarity pattern, but they were uncommon in our samples. Endpoint difference was the only index that did not differentiate among the treatments in spring 1998, therefore suggesting that high variance in species density affected the treatment comparisons. The common leafhopper *J. olitoria* was the sole indicator shared by proportional similarity and the Bray-Curtis-based index, thus replacing *Sminthurus* sp. 1, which was not a strong contributor to any of the indices.

All three indices showed significant treatment differences in the following two seasons. In fall 1998, the leafhoppers (*Empoasca* spp.) were the group to contribute significantly to more than one index (proportional similarity and endpoint difference). These species' response to all hardwood reduction methods, especially fire, was pronounced as virtually no individuals were ever recorded from control plots (Figure 3). As in the previous season, no species significantly influenced all three indices during spring 1999. *Sminthurus* sp. 1, the jumping spider #25 (unidentified), the planthopper *Oecleus* sp., and the ant *Crematogaster ashmeadi* Mayr were the only indicators identified by at least two indices. Because the restoration effect for endpoint difference was barely significant ($P = 0.046$), jumping spider #25, *Oecleus* sp., and *C. ashmeadi* may not be the most reliable indicators of restoration success. This left *Sminthurus* sp. 1 as the strongest potential indicator, achieving its highest densities in felling/girdling plots, somewhat lower in herbicide and burn plots, and the lowest densities in control plots (Figure 3).

Overall, very few species dominated the list that we would suggest as indicators of restoration success over time: *Sminthurus* sp. 1, *J. olitoria*, *Empoasca* spp., *Erythroneura* spp., and *M. pruinosa*. With the exception of *Empoasca* spp., these species were generally common and statistical analyses could be performed on their densities (see Figure 3). Clearly, there was a connection between fire and the increased densities of these arthropods: we hypothesize that the palatability of resprouting plants is the main cause of this interaction.

Management Implications

- Long-term management of sandhills and other communities on Eglin on a large scale requires economical methods that stimulate productivity in the understory. Only fire will satisfy these two constraints for herb-layer arthropods.

Prescribed burning is substantially cheaper to apply than herbicide and felling/girdling. In 2000, burns started with ground ignition cost approximately $21.90/ha, whereas aerial ignition can now be accomplished for $8.60/ha (J. Furman, EAFB, pers. comm.). Herbicide application operations amount to $140.80/ha for the herbicide and $76.57/ha for labor (total $217.40/ha). Chainsaw felling/girdling is $158.10/ha, involving only labor. These estimates do not include the cost of a fuel reduction burn that would normally follow herbicide and felling/girdling operations.

- The relationship between fire, understory arthropods, and the diet of red-cockaded woodpeckers requires further research as this connection affects their management and that of other species in longleaf pine forests.

The diet of red-cockaded woodpeckers during the breeding season depends exclusively on arthropods. Hanula and Franzreb (1998) have shown that 70

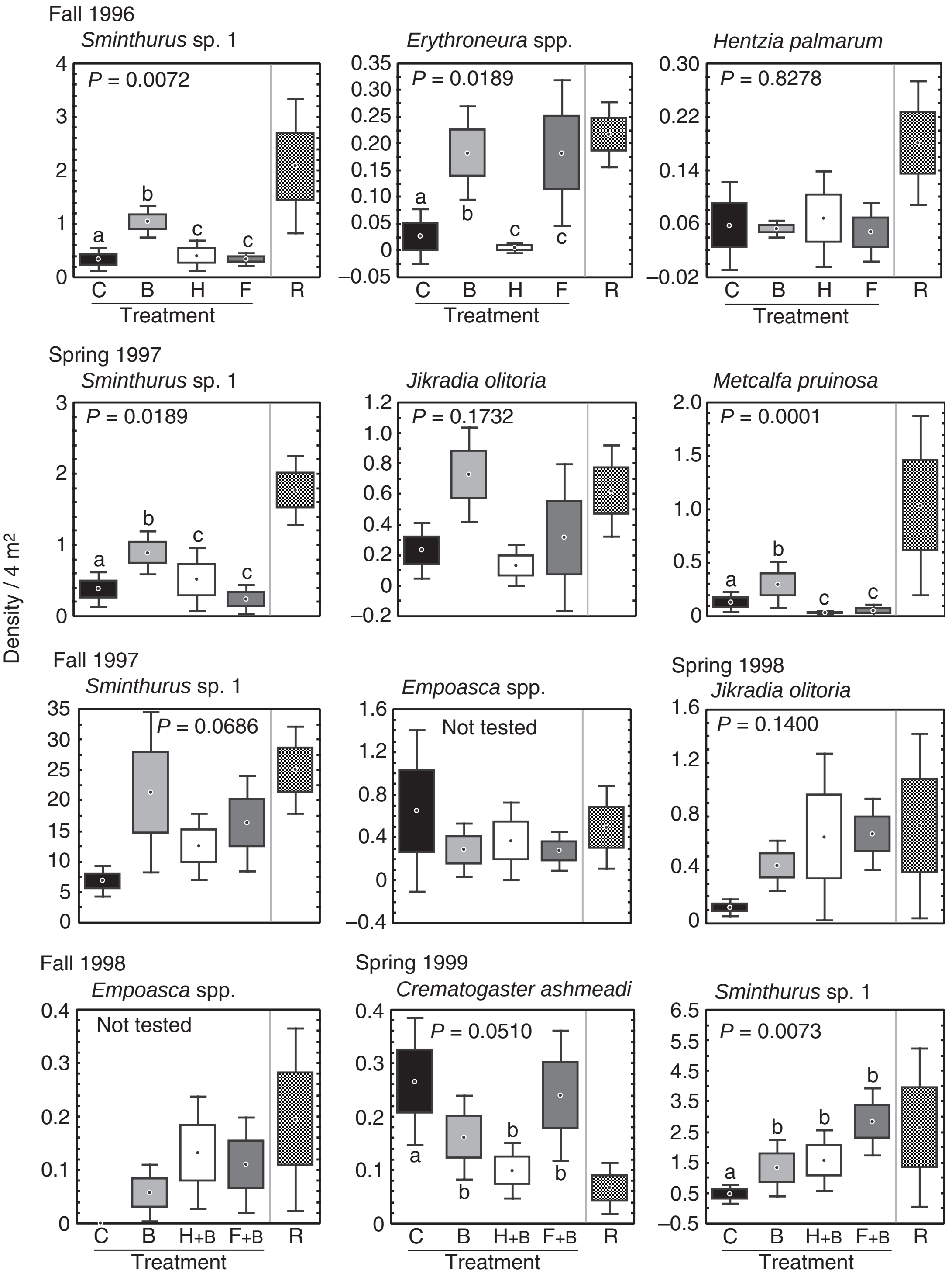

Figure 3.—The densities of herb-layer arthropods identified as the strongest indicators of restoration success by similarity analyses in hardwood reduction and reference plots at Eglin Air Force Base, Florida, from fall 1996 to spring 1999. Indicators were only selected when similarity values were significantly different among hardwood reduction treatments. Tests of treatment effects were calculated with two-way ANCOVA. The experimental design is a randomized complete block (6 blocks), split-plot design, but only the block design at the whole plot level is presented here. The covariate was the pre-treatment data from the fall 1994 or spring 1995. The error term is the mean square of the interaction of the block and restoration treatment effects. Density values received various transformations when it was necessary to stabilize variances. The center of the box is the mean, the edges of the box correspond to one standard error, and the error bars are a 95 percent confidence interval.

percent of the arthropod prey consumed by red-cockaded woodpeckers below the crown of longleaf pines disperse from the understory (mostly ground/soil arthropods). Therefore, there appears to be a strong dietary link between fire, understory arthropod population growth, and the woodpeckers' reproductive success (see also James et al. 1997).

- Little is known about arthropods in second- and old-growth longleaf pine forests. Quantitative and taxonomic research on this subject would greatly help both ecologists studying other aspects of longleaf pine forest ecology and managers needing to conserve arthropod communities and recover the populations of threatened and endangered species.

We identified potential indicators to monitor that may inform managers on longleaf pine sandhill quality and fire intervals. It is worthwhile to determine if decreasing similarity of herb-layer arthropods between managed and reference sites may roughly correspond with the end of a fire-free interval.

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Responses of Amphibians to Fire Disturbance in Pacific Northwest Forests: a Review

R. Bruce Bury[1], Donald J. Major[2] and David Pilliod[3]

Abstract.—In western North America, major wildfires often now result in stand-replacement events and natural resources losses for many decades post-burn. Fire severity has been exacerbated by past fire suppression that has allowed large fuel load accumulations. To reduce woody debris and restore the ecological integrity of western forests, prescribed burning is increasingly used as a regional management tool. However, we do not understand the effects of either wildfire or prescribed fires on amphibians in stream, riparian and terrestrial habitats in western forests. Terrestrial amphibians, macroinvertebrates and other animals are surface active during periods of rainfall or high moisture. Wildland fire usually starts in the hot, dry summers typical of these more arid Western and Mediterranean climates and may have less effect on resident biota than prescribed fires often conducted during the late fall to spring rainy season, when there is sufficient moisture to prevent crown fires. Still, intense wildfires may result in increased erosion and sediment or changes in soil chemistry impacting downstream aquatic environments. To our knowledge, no published reports exist on effects of fire on the aquatic herpetofauna of the Pacific Northwest. Research efforts now underway include new studies of wildland fires in Oregon and Idaho on aquatic amphibians, and studies on the effects of prescribed fire on terrestrial salamanders and associated forests in the Klamath Province along the Oregon-California border. These will help evaluate the cumulative effects of fuels reduction on amphibian population and habitat structure, and provide guidelines to better manage for wildlife species characteristic of western forests. In the Pacific Northwest, investigations of fire effects on wildlife are severely lacking relative to the vast acreage, economic value, and biodiversity of its forest ecosystems. Given the increasing prominence of wildfire and prescribed burning in many western forest systems, we suggest more resources will be devoted to such research endeavors, and that they include other sensitive groups of wildlife such as mollusks.

Fire is a natural, recurring disturbance in forested ecosystems of western North America, but it has been aggressively suppressed for >50 years. Although fire prevention was implemented to protect forest resources, these efforts have resulted in greatly increased fuel levels and, in turn, increased risk of catastrophic fires (Pyne 1982, Agee 1988, Henjum et al. 1994). Catastrophic fires are generally defined as stand-replacing fires that burn at spatial scales and intensities atypical to the historic fire regime. For example, forested environments with long fire return intervals (300-600+ years) are subject to large-scale stand replacing fires because fuel loadings are typically high (Morrison and Swanson 1990). Of greater importance might be how past forest management activities such as clearcutting and thinning and associated forest fragmentation affects the resulting forest structure and the fire mosaic when they do burn. In contrast, forests with historically short fire return intervals (20-40 years) were characterized by reduced fuel loads where stand replacing fires were less common. Fire suppression is hypothesized to have the greatest affect on forests with short (i.e., <50 year) fire return intervals.

Resource managers now recognize the importance of fire for maintaining healthy forests, yet often face conflicting priorities when pursuing multiple resource management objectives. Prescribed burning, prescribed wildland fire (e.g., allowing natural fires to burn within specified parameters), and other fuels management practices are being introduced into the landscape to reduce fuel loadings, but little is known about their effects on the biota of the forests that are being restored or altered (Potter and Kessell 1980). For example, we lack information on the quantity and quality of the resulting downed woody debris that comprises the critical habitat for many species of resident wildlife. Current standards and guidelines of the Northwest Forest Plan require Federal land managers to promote retention of dead-and-down wood as wildlife habitat yet reduce high fuel loads through prescribed burning or other fuels reduction practices (USDA/USDI 1994).

Further, we lack information on both the short- and long-term suitability of the post-burn habitat for resident and migratory wildlife. Fuels management will alter the structure and composition of existing fire-suppressed systems. Large wood plays a vital role in ecosystem processes such as nutrient and water cycling (Harmon et al. 1986, Franklin and Spies 1991). It provides a moist, thermally stable habitat for many species of wildlife both large and small (Maser and Trappe 1984, Carey and Johnson 1995, Bull et al. 1997). We lack basic information on the response of large woody debris to burning and the suitability of fire-scarred woody debris for wildlife.

In Pacific Northwest forests, terrestrial salamanders are strongly associated with structural elements of the forest

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[1] USGS Biological Resources Division, Corvallis, OR 97331
[2] Department of Fisheries and Wildlife, Logan, UT 84321
[3] Aldo Leopold Wilderness Research Institute, University of Montana, Missoula, MT 59807

floor such as woody debris, moss, and surface/subsurface rock (Bury and Corn 1988, Corn and Bury1990, Welsh and Lind 1995). Stream amphibians also are dependent on the structural components of stream substrate including in-channel large woody debris (Corn and Bury 1989). However, almost nothing is known about the responses of amphibians and their habitats to fire and fuels management practices in the West (DeMaynadier and Hunter 1995, Welsh and Droege 2001). Only a few unpublished studies exist on the effects of fire on pond-breeding amphibians in western forests exist (Askey and Peterson 1993). These limited data sets indicate no significant differences in amphibian occurrence in ponds located in burned versus unburned forests.

There are some recent studies on the effects of prescribed fire on herpetofauna in the eastern U.S. (Russell et al. 1999, Ford et al. 1999), where fire frequency has been reduced and the season of burn changed from mostly growing-season to dormant-season in Southeastern pine forests (Means and Campbell 1980, Robbins and Myers 1992). Widespread public and agency concerns over large fires are now driving policy-level decisions to integrate fire and other fuel reduction strategies into management of ecosystems.

Our goal is to review and compare the effects of fire and fuel reduction management on Pacific Northwest amphibians and their associated habitats, especially downed woody material. We will attempt to summarize: (1) what we know generally from a few available studies, and (2) what information is needed.

Stream Amphibians

There are three families (10 species) of amphibians endemic to the rocky, cool-water streams in the mountainous regions of the Pacific Northwest. Many of these species are of conservation concern due to declining populations (Bury 1994, Smith et al. 1998). Therefore, understanding fire effects on stream amphibians and their habitats will be of utmost importance in future management practices in Northwest forests. Some stream amphibians such as the giant salamander (*Dicamptodon* spp.) may respond positively to increased stream productivity (increased periphyton or algae growth) for a few years after forest opening from timber harvest (Murphy and Hall 1981) and presumably fire. Stream amphibians also are sensitive to changes in debris flow, elevated temperature, water chemistry, and sedimentation that often occur post-disturbance (Bury 1988, Gamradt and Kats 1997, Kerby and Kats 1998, Welsh and Ollivier 1998). For example, the tailed frogs (*Ascaphus* spp.) are among the least tolerant of the anurans to elevated temperature and will die quickly when exposed to water temperatures at or near 29.6°C (deVlaming and Bury 1970). In contrast, giant salamanders tend to have broader tolerance to temperature and siltation than other stream amphibians (Bury 1988).

Although direct mortality of aquatic and wetland-associated life stages of amphibians may be low where wet areas and riparian zones provide refugia from fire (Vogl 1973), terrestrial life stages in nearby uplands may experience much higher mortality associated with the direct and indirect effects of fire that alter prey availability or change shelter and microclimate (Lyon et al. 1978, 2000; Russell et al. 1999). Further, physical and biological changes in adjacent uplands may influence the survival and well-being of biota in riparian zones through downslope effects on hydrology and water chemistry in streams (Minshall 1989).

Some research on large, stand-replacing wildfires on lotic biota suggests that fire ultimately benefits aquatic invertebrates and fishes, even those species that are negatively affected by the disturbance immediately after the fire (Roby and Azuma 1995, Minshall et al. 1997, Rieman et al. 1997). For example, Lyon et al. (1978) suggested that some aquatic invertebrates may decline immediately after a fire, then increase to levels above pre-fire conditions as a response to increased stream productivity. Large fires can have long-term effects on streams by: (1) reducing invertebrate diversity for a decade or longer (Roby and Azuma 1995); (2) changes in peak discharge, stream channel morphology, large woody debris inputs, and sediment loadings; and (3) elevated temperature and altered water chemistry (Richards and Minshall, 1992; Minshall et al. 1997). Also, the effects of fire on stream biota may be more pronounced in headwater streams than in mid-order or larger streams (Roby and Azuma 1995, Minshall et al. 1989, Minshall et al. 1997).

Although there is little empirical evidence of how fire impacts stream amphibians, we can glean some information from the Old-growth Wildlife Habitat Program in the Pacific Northwest (Ruggerio et al. 1991), which compared biota across chronosequences of forest categorized into young, mature, and old-growth stands. Because almost all of these stands were naturally regenerated from wildfires, the younger stands were most recently burned. Thus, fire-sensitive wildlife populations would be expected to be different in younger stands compared to older, mature stands that had a longer time since disturbance.

Spies (1991) estimated the age of trees across the chronosequence of stands in the Oregon Coast Range (Table 1) and reported that the young and mature stands were not the equivalent of intensively-managed plantations resulting from harvest. For example, there often were large amounts of downed woody debris in young natural stands, resulting from input of fallen trees that were fire killed. There was less wood volume in mature stands and then large accumulations again in old-growth. Managed stands tended to have low

Table 1.—Estimated ages of trees in stands in the Oregon Coast Range (from Spies 1991).

Stand type	Young	Mature	Old-growth
Mean age	55	100	315
Range	30-79	84-120	130-525

amounts of downed wood once slash rotted or was burned on site. Clearcut stands lack large trees and, in turn, recruitment of large downed woody material on the forest floor.

Bury et al. (1991a) found no significant difference among stream amphibians across natural young, mature, and old-growth stands in three Northwest biogeographic provinces (Table 2). However, Corn and Bury (1989) reported that the density and biomass of four species of stream-associated amphibians were significantly greater (2-7X) in uncut forest (natural regeneration) than in streams in clearcut stands sampled 14-40 yr post harvest. Although there were few young natural stands (*n* = 3), the abundance of amphibians was similar to streams found in mature and old-growth stands (Table 3), but had markedly greater numbers compared to streams running through clearcut stands. This evidence suggests that wildland fire had little effect on stream amphibians or amphibians recovered rapidly in postburn conditions, but clearcut logging was detrimental and had long lasting effects. However, further study is needed on this topic, especially with more replicates of study streams.

Studies on prescribed and wildland fire effects on stream-breeding amphibians in the Northwest are currently underway. In Idaho, we (Pilliod and P. S. Corn) have initiated a 3-year study comparing Rocky Mountain tailed frog (*Ascaphus montanus*) populations in streams running through both burned and unburned forests. In conjunction with this retrospective study, we are conducting an experimental prescribed fire study in the South Fork Salmon sub-basin where Rocky Mountain tailed frog and Idaho Giant salamander (*Dicamptodon aterrimus*) populations will be monitored in six streams for 3 years pre- and 2 years post-burn. Similar research will start in Oregon in 2002 by one of us (Bury).

Riparian Habitats

Riparian zones adjacent to streams are important habitats for Pacific Northwest amphibians (Bury 1988). Most of the endemic stream amphibians of the region inhabit the waters and adjacent cool, vegetated banks. Riparian vegetation serves to shade and cool stream temperatures, and protect the rocky substrate from siltation from upslope disturbance. Stream amphibians like tailed frogs (*Ascaphus truei*) often move into the riparian zone after transformation (T. Wahbe, pers. comm.), where conditions are relatively cool and moist year-round compared to hotter, drier upslope areas.

In forested stands west of the Cascade crest, riparian areas associated with permanent streams are not scheduled for prescribed burning (T. Atzet and J. Lint, pers. comm.). These Federal lands have been protected as riparian reserves under the Northwest Forest Plan (NWFP). The Aquatic Conservation Strategy of the NWFP provides riparian reserves of 1-2 tree heights away from all permanent streams. In 1997, an unpublished NWFP report on "Riparian Reserve Evaluation of Techniques and Censuses: Federal Guide for Watershed Analysis" suggested that management activities in riparian areas are feasible, including fuels reduction. Introduction of prescribed fire into riparian areas appears to be only allowable on an experimental basis for research or pilot studies at this time. However, prescribed fire likely will occur near riparian zones and there may be an influence from these adjacent, upslope areas.

Fire suppression in riparian areas for long periods will create fuel build-ups that may eventually lead to unnaturally severe fire in riparian zones, especially in the more productive sites at lower elevations. Further, disruption of the natural fire regime likely will result in changes in riparian forest structure and composition. This may include increased amounts of input of woody debris into streams that, in turn, influence channel processes and habitat availability for stream or streamside amphibians.

Terrestrial Habitats

To our knowledge, there are no studies addressing the effects of fires on terrestrial amphibians in the Pacific Northwest. However, we can again glean useful information from the Old-growth Wildlife Habitat Program (Ruggerio et al. 1991). Bury et al. (1991b) summarized results for 130 stands in the chronosequence from three Provinces (OR Cascades, OR Coast Range, WA Cascades). Each stand was sampled in the fall for one month of pitfall trapping (1,080 trap-nights/stand), and most were sampled twice (1983 and 1984). This intensive effort showed that there were no significant differences in catch of 6 of 8 common species of amphibians along the chronosequence with young stands did not differing from mature or old-growth. The Northwestern salamander (*Ambystoma gracile*) was more common in old-growth whereas red-backed salamanders (*Plethodon vehiculum*) were more frequent in young stands than in other types. However, both of these species may be influenced by factors other than stand age. The Northwestern salamander is a migratory species that travels overland and location of breeding ponds may greatly influence its local abundance. The red-backed salamander requires rocky substrate and this may have influenced its distribution. Overall, the

Table 2.—Distribution of 79 streams by stand category and natural regenerated or harvested state.

	Natural Regeneration			Clearcut Harvest
	Young	Mature	Old-growth	
Washington Cascades	6	6	6	0
Oregon Cascade	6	6	6	0
Oregon Coast Range	3	10	10	20

Table 3.—Stream-associated amphibians taken from three categories of forest stands in the Oregon Coast Range. Results are mean numbers caught/10 m (most were headwaters about 1 m wide).

Species	Old-growth and Mature (n = 20)	Young Natural (n = 3)	Young Harvest (n = 20)
Tailed frog, *Ascaphus truei*	9.8	8.3	3.5
Torrent salamander, *Rhyacotriton variegatus*	3.1	1.7	0.4
Pacific giant sal., *Dicamptodon tenebrosus*	23.4	34.3	4.6
Dunn's salamander, *Plethodon dunni*	4.6	3.0	1.4

resident amphibians were either little influenced by fire or had recovered since burning 3-5 decades earlier.

Using pitfall trapping in the Oregon Coast Range, Corn and Bury (1991) found no marked differences in numbers of the five most common amphibians across the chronosequence of natural regenerated stands (from earlier fires): young, 40-75 yrs old (n = 8 stands); mature, 80-120 (n = 10); and old-growth, 150-450 (n = 27). They also sampled 5 clearcut harvested stands (< 10 yrs post harvest) and overall abundance did not differ much from results in natural stands (young to old-growth) except that tailed frogs and torrent salamander (*Rhyacotriton* spp.) were absent. Both groups appear associated with streams and both are highly sensitive to logging.

Employing time-constrained searches of downed woody debris, Corn and Bury (1989) found a correlation between ensatina (*Ensatina eschscholtzi*) and clouded salamander (*Aneides ferreus*) numbers and stand amounts of downed wood. Loss of large wood input was hypothesized to be a limiting factor for several species of terrestrial plethodontids in Northwestern coniferous forests.

In northern California, Welsh and Lind (1991) reported that more species and numbers of individuals of amphibians, especially terrestrial salamanders, were in older forests compared to younger stands. These also were in the naturally regenerated chronosequence. These authors and others (Bury 1983, 1994; Smith et al. 1999; Welsh and Droege 2001) suggest that timber harvest has negative effects on several species of terrestrial salamanders in western coniferous forests. Similarly, there appear to be reduced numbers of salamanders in logged stands in deciduous and mixed forests in the eastern U.S. (Pough et al. 1987, Petranka et al. 1993, deMaynadier and Hunter 1995). Timber harvest, especially clearcutting, opens up forest canopies and leads to desiccation. Removal of trees reduces input of large chunks of woody debris into the forest ecosystem.

Further, harvested areas oftern are often subjected to secondary site preparation or other pre-commercial treatments such as burning, herbicide spraying and thinning to enhance new tree growth. Although we know little about how such multiple stressors work in concert, timber harvest (especially clearcutting) and associated silvicultural practices appear detrimental to terrestrial amphibian populations (De Maydanier and Hunter 1995, Welsh and Droege 2001).

To address how fire influences forest wildlife, we (Bury and Major) are involved with a study of fire effects on terrestrial amphibians in the Klamath Province (southern Oregon and northern California). Key objectives are to:

(1) Compare structural components of the forest floor and their use by terrestrial herpetofauna in burned and unburned sites;

(2) Determine the vulnerability of the structural components used by herpetofauna; and

(3) Evaluate habitat quality by relating diversity and abundance of herpetofauna to available forest floor structure.

Our study has two designs: retrospective (wildland fire) and experimental (prescribed fire with pre- and post-treatment). Retrospective work will describe forest floor structure in burned and adjacent unburned sites in recent (< 10 yr) wildland fires. We will conduct time-constrained terrestrial surveys at the paired sites to compare species richness and relative abundance of herpetofauna, and to characterize the use of the structural components by amphibians.

Our preliminary results from one large wildland fire in the North Umpqua River Basin in Oregon in 1996 indicate that there was no negative effect from this fire on terrestrial herpetofauna. We found more individuals in the burned than unburned comparison areas using 3 paired plots. However, much more cover was available in the unburned stands, so catch per cover object is somewhat less in the burned forest. Lack of response by the resident terrestrial amphibians may be related to occurrence of this wildland fire in summer during normally dry, hot conditions when terrestrial species of amphibians are deep underground. However, lack of information regarding pre-wildfire conditions (i.e., fuel loadings, cover availability, and associated amphibian detection rates) limits our inference capabilities. We have no data on possible long-term effects.

Unlike most wildland fires, prescribed burning activities often coincide with seasonal surface activity periods of terrestrial salamanders in the spring and fall. We (Major and Bury) are currently examining the effects of prescribed fire on herpetofauna in late seral coniferous and mixed-coniferous forests in the Klamath Province. Specifically, we are examining pre- and post-burn population-level responses of terrestrial salamanders across a series of prescribed fires. We also will compare structural components of the forest floor such as fire fuels and salamander habitat. Finally, we hope to explore the effectiveness of modeling population-level responses of terrestrial salamanders to changes in forest floor "habitats" through the use of predictive fuels consumption and fire effects modeling. Field sampling began in Summer 2001 with project completion expected in 2003.

Discussion

The lack of information on the effects of fire on fish and wildlife is a major impediment to developing ecologically sound fire management policies (McMahon and deCalesta 1990). Recent reviews of the effects of fire on amphibian communities uncovered relatively few studies, although most of the work reviewed had occurred in Southeastern pine plantations (Russell et al. 1999). Most aquatic studies have focused on stand-replacement burns and few have included an experimental component, such as prescribed burns (Major and Bury 2000). Further, many recent studies have examined population descriptors such as species presence or relative abundance, whereas more specific information on population change and measures of productivity may be needed to understanding cause-and-effect relationships of fire in the Pacific Northwest.

Although recent directives promote fuels reduction, we lack information on the quality of resulting postburn habitats or how resident wildlife species respond to fire-induced changes in availability of the altered structural elements (e.g., less forest duff, fewer cover objects). Reduction in quantity of fuels, downed woody debris, also could have a profound effect on the habitat needs of wildlife and associated biological integrity that federal land management is charged to conserve. Overall, scientific studies on fire effects on wildlife and associated habitats are just starting in the Pacific Northwest. The current studies will provide some information for evaluating the effects of fire and forest management practices on several amphibian species in stream, riparian, and terrestrial situations.

We believe there is a need for more investigations that relate fuels management to habitat quality for resident terrestrial herpetofauna for several reasons. Many terrestrial amphibians are sensitive to habitat change because they have moist, permeable skin and restricted home ranges. Most terrestrial amphibians and some reptiles also require large woody debris as nesting habitat or cover objects (Welsh and Droege 2001). Thus, herpetofauna are well suited as resident wildlife to measure responses to habitat changes.

There is a need to assess prescribed fires inside riparian reserves and adjacent slopes. For example, unburned riparian areas likely buffer the stream from the effects of fire immediately after the burn, but streams may have a delayed response until spring runoff carries sediment and nutrients into the stream during periods of peak discharge. On longer time scales, excluding burning in riparian areas may result in increased riparian forest and amounts of instream woody debris dynamics. This may increase amount of woody debris and pose greater fire risk in riparian areas, especially during drought periods. Amphibians may still respond to upland burning, because juveniles and adults of stream amphibians move into adjacent woods at the onset of fall rains and fire-associated mortality of terrestrial phases of these species may occur in prescribed fire periods.

The few studies of fire effects on streams to date have been conducted within single drainages or fire complexes and have been focused on stand-replacing wildfires. No published studies exist that address the relative effects of stand-replacing and prescribed fire at broad spatial scales. There is need for studies that combine an ongoing prescribed fire study with post-fire investigations of stream response to widespread wildland fires. These will provide opportunities to characterize the effects of fire treatments and severity on aquatic and riparian habitats.

We need integrated regional and national programs that better link wildlife habitat components into fuels inventories and fire effects models. Although inventory techniques are designed to collect information on structural components pertinent to fuels management goals, this information is of limited use in evaluating "habitat use" by wildlife. Further, a habitat/fuels approach would prove biologically useful in developing effective monitoring protocols and providing baseline information to elucidate working hypotheses on fire effects and ecological responses across different spatial scales. In the western U.S., much work has focused on inventorying volume and tonnage of downed woody fuels (Koski and Fischer 1979, Brown et. al. 1982). However, these studies address only a subset of structural components and do not provide information on fire-mediated changes in quantity, type, size, and physical characteristics of downed wood, which affects wildlife habitat quality (Bull et. al. 1997). Fire effects models (Peterson and Ryan 1986, Ryan and Reinhardt 1988, Keane et al. 1994) incorporate a wider range of fuel types and more precise description of post-burn conditions; however, they are limited in their utility to describe habitat quality across the range of structural legacy in northwest forests. Furthermore, these models only examine immediate post-fire changes and are based on modeling structural components influenced by past suppression.

Clearly, we should increase numbers and integration of projects that: 1) Determine what type and amount of surface and ground structure are required by resident wildlife, 2) Evaluate direct and indirect effects of fuels management on structural components important to resident wildlife, and 3) Predict the short- and long-term effects of fuels management on legacy components in forests and associated quantity and quality of habitat for resident wildlife. There is a rich literature and growing understanding of habitat requirements of forest wildlife, in part related to major efforts generated by the Northwest Forest Plan and other agency mandates. A future goal is to tie this information to fuels inventory models and integration of disciplines. Lastly, we need to recognize the vast acreage and diversity of forests in western North America, Geographically dispersed studies will be required to better define how wildland and prescribed fires affect wildlife across a broad spectrum of environmental conditions.

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The Effects of Burning and Grazing on Survival, Home Range, and Prey Dynamics of the Texas Horned Lizard in a Thornscrub Ecosystem

Anna L. Burrow[1], Richard T. Kazmaier[1], Eric C. Hellgren[1], and Donald C. Ruthven, III[2]

Abstract.—We examined the effects of rotational livestock grazing and prescribed winter burning on the state threatened Texas horned lizard, *Phrynosoma cornutum*, by comparing home range sizes, survival estimates and prey abundance across burning and grazing treatments in southern Texas. Adult lizards were fitted with backpacks carrying radio transmitters and relocated daily. Prey abundance (harvester ants, *Pogonomyrmex rugosus*) and activity were greater in burned pastures, but grazing had a variable effect depending on the timing since the last burn. Home ranges in burned pastures were smaller than in unburned pastures in the active season. Level of grazing (heavy vs. moderate) did not affect home range size. Summer survival rates of horned lizards were higher in the moderately grazed sites than the heavily grazed sites. The smaller home ranges, lack of effect on survival rates, and greater prey abundance in burned pastures suggested a positive effect of fire on Texas horned lizards.

Introduction

The effect of land-use practices on sensitive species, such as threatened or endangered species, is of considerable conservation and political interest. However, little information is available to evaluate the ecological effects of management practices such as burning or grazing on herpetofauna in general (Russell et al. 1999), and on the threatened Texas horned lizard, *Phrynosoma cornutum*, in particular. The Texas horned lizard is the official state reptile of Texas (Donaldson et al. 1994), and is a species of special concern in the conservation community. Although the Texas horned lizard was protected by Texas legislative mandate in 1967, it has declined throughout its range, especially in Texas (Price 1990). Suggested reasons for this decline include habitat alteration for land uses such as agriculture or development, the introduction of the red imported fire ant (*Solenopsis invicta*), and the use of insecticides (Price 1990; Donaldson et al. 1994). Such declines can decrease genetic variability and hinder the lizard's ability to adjust to changes in environmental conditions caused by land-use practices.

The habitat and prey of horned lizards can potentially improve or worsen with fire and grazing. Fire reduces shrub canopy cover (Dunne et al. 1991) where grazing can increase the amount of woody vegetation (Archer and Smeins 1991). Ruthven et al. (2000) found that forbs increased on southern Texas rangelands in the first year after a winter burn, but were not affected by grazing. Bunting and Wright (1977) also found that forbs and grass cover increased following fire. Ants, the main prey of horned lizards, are not deleteriously affected by fire or grazing (McCoy and Kaiser 1990; Heske and Campbell 1991; McClaran and Van Devender 1995:165 Fox et al. 1996).

Specific objectives of our research were to: compare the relative abundance and activity of harvester ants, *Pogonomyrmex rugosus*, the main food source of the Texas horned lizard, among different burning and grazing treatments; and compare home range size and survival rates of Texas horned lizards among different burning and grazing treatments. Based on available literature, we made several testable predictions. Harvester ant activity and abundance should be greatest in the moderately grazed/burned site because of an increase in seed production coupled with an open, sparsely vegetated habitat, selected by harvester ants (DeMers 1993). Horned lizards will be less selective of foraging habitat in the moderately grazed and burned site due to greater prey abundance and better habitat interspersion. Therefore, we predicted range size of Texas horned lizards would be smaller and survival rates higher in the moderately grazed/burned sites than in other treatments.

Study Area

The 6,150-ha Chaparral Wildlife Management Area (CWMA) occurs in Dimmit and La Salle Counties, Texas. The CWMA was purchased by the state in 1969 and management authority was given to the Texas Parks and Wildlife Department (TPWD). Average annual rainfall on CWMA is 63 cm with a primary peak in May and a secondary peak in late September/early October (TPWD, unpublished data). The dominant vegetation types on the CWMA are honey mesquite (*Prosopis glandulosa*) woodlands or parklands, with prickly pear cactus (*Opuntia engelmannii*), tasajillo (*Opuntia leptocaulis*), brasil (*Condalia hookeri*), spiny hackberry (*Celtis pallida*), blackbrush acacia (*Acacia rigidula*), twisted acacia (*Acacia schaffneri*), hogplum (*Colubrina texenis*), and Texas persimmon (*Diospyros texana*) as common subdominants. Common and scientific names for vegetation follows Hatch et al. (1990).

Five study sites (50-60 ha) were selected on the CWMA, each with a different burning and grazing treatment.

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[1] Department of Zoology, Oklahoma State University, Stillwater, OK 74078

[2] Texas Parks and Wildlife Department, Artesia Wells, TX 78001

Sites were chosen based on similarities in dominant woody species and canopy coverages. Treatments were: control (non-burned/non-grazed), moderately grazed/ burned, heavily grazed/burned, moderately grazed/non-burned, and heavily grazed/ burned. The control site has not been burned or grazed since 1976.

Historical grazing occurred on CWMA, but after TPWD began managing the land, grazing steadily declined and temporarily stopped in 1984 because of poor range condition. During this time, the grazing system was changed from continuous grazing to different rotation systems. Grazing resumed in 1991 with a high-intensity, low-frequency rest-rotational system from 1 October to 30 April. Moderately grazed areas were stocked at 25 animal-unit days (AUD) • ha^{-1} • yr^{-1} and heavily grazed areas were stocked at 37.5 - 50 AUD • ha^{-1} • yr^{-1}. We defined one AUD as 2 steers for one day.

A prescribed burning program was initiated on the CWMA in 1997. Burns were conducted using head fires ignited with a drip torch and covered 40 to 80 ha. The study areas used in this research project were burned in February 1998 and November 1999 during dry conditions.

Methods

Field Methods

Lizards were captured in each of the study areas through road cruising, fortuitous encounters, and drift fence arrays. Each study site (*n*=5) on the study area had 3, Y-shaped drift fence arrays that were open for 14 days in either May or June. Upon capture, snout-vent length (SVL), total length, mass, sex, and location of the lizards were recorded. Lizards were marked with an intra-abdominal passive integrated transponder (PIT; AVID, Norco, California, USA) tag. The fifth toe on the front right foot was also clipped to recognize if the lizard had been previously caught. Lizards that were too small to receive a PIT tag, approximately < 50 mm SVL, were given a unique toe clipping sequence.

Adult lizards captured within the five study sites were fitted with custom-made backpacks that carried transmitters (150-151 MHz, L and L Electronics, Mahomet, Illinois, USA). Backpacks were composed of a beige muslin material and elastic straps dyed to match the natural substrate color of the CWMA. The backpack was attached to the lizard by placing an upper strap around the neck and one front leg, and placing an additional strap around the back legs. A drop of cyanoacrelate gel adhesive was used to attach the straps to the lizard's chest and lower abdomen to further secure the backpack. The total mass of the transmitter and backpack bundle was approximately 3 g (< 8 percent of lizard mass). Receiving range of the transmitters was approximately 100 m. An antenna attached to the end of a 5-m PVC pole increased transmitter detectability to approximately 200 m.

Radio-fitted lizards were initially relocated twice daily with a handheld two-element Yagi antenna until lizards resumed normal ranging behavior. Monitoring was then reduced to once daily until hibernation. Every six weeks, lizards were recaptured and given a new transmitter in the field. Once refitted with a backpack, the lizard was released. Data recorded at each relocation included lizard activity and behavior, date, time of day, pasture, burn treatment, UTM coordinates, weather, and micro-habitat data.

Data were collected during the summers of 1998-2000. The summer was divided into 2 seasons, active and inactive, corresponding to the relative activity of horned lizards. The season encompassing 15 April - 30 June was the time of greatest lizard activity and was considered the active season. Lizards are considerably more sedentary during 1 July - 15 August, which was termed the inactive season.

Locations of lizards were estimated by pacing from the position of the lizard in a cardinal direction to a road and then to a permanent landmark with known Universal Transverse Mercator (UTM) coordinates. Coordinates were entered into a Geographical Information System (GIS) to aid in range and habitat analyses. Only lizards with a radiotransmitter were used in home range and survival rate analyses (Munger 1986). All statistical analyses were considered significant at a = 0.10.

Ant Abundance and Activity

Ant abundance and activity were measured with bait stations composed of six petri dishes placed 15-m apart along a transect. Transects were randomly located and followed a compass bearing. Each bait station was baited with millet and was anchored to the ground with a nail to prevent rodents from removing the dishes. Four transects from the same study area were run simultaneously. Petri dishes were baited in the morning and checked between 0800 and 1100 for ant activity to encompass the peak activity of ants (Whitford and Bryant 1979). Number of ants foraging at the station and the number of ants visiting the station within one minute were recorded. Though other species of ants were noted if present, only harvester ants were counted. The bait stations were baited again in the evening to assure that ants would keep visiting the dishes. Bait stations were run for 4 days at a time using the same transects once in the active season and once in the inactive season in all five study areas in the summers of 1999 and 2000. Systematic searches for ant mounds (Whiting et al. 1993; Fair and Henke 1997) were not used in this study because it was difficult to determine harvester ant mounds from other ant mounds, and to distinguish active ant mounds from inactive ant mounds.

Harvester ant abundance was averaged across the 4 days for each transect, providing 4 replicates for each study site. Differences in ant activity and abundance were compared across the different burning and grazing treatments using a repeated measures, 4-way ANOVA including burning (burn, unburn), grazing (moderate, heavy), year (1999, 2000), and season (active, inactive) as main effects and all interactions. The treatments creating the repeated measurements were season and year. Because of a small sample of lizards in the control site (ungrazed, unburned), and the lack of an ungrazed, burned site, the control site was not used in the ANOVA. Instead, we examined the following contrasts: control vs. grazed sites, control vs. the grazed, unburned sites, and control vs. the heavily grazed, burned site (the most disturbed site).

Home Ranges

Range size of lizards were calculated using 95 percent minimum convex polygons (MCP; Mohr 1947) with the Animal Movement Analysis Program (Hooge et al. 1999). We included lizards tracked for ³ 20 locations to ensure a reliable representation of the home range. Because the home range data were not normally distributed, home range sizes were log-transformed. Individual lizards were used as the experimental unit, although this represents pseudoreplication, because the treatments were not replicated (Hurlbert 1984). Therefore, any inferences made from these data should be used with caution beyond the study area.

Comparisons of range size were made with a 3-way ANOVA including burning, grazing, and season as main effects and all interactions. Preliminary analysis indicated no gender differences in range size. Therefore, data were pooled across sex. Contrasts to the control were calculated in the same manner as described above in the ant analyses.

Survival

Survival rates were estimated using the Kaplan-Meier procedure (Pollock et al. 1989). Only lizards tracked for ³ 10 days were used in these analyses. The fate of many lizards was unknown due to transmitter failure, removal by a predator, lizard migration, and discovery of a backpack (without a lizard). Lizards with an unknown fate were termed censored in the analyses. To determine survival, we estimated the fate for censored lizards based on knowledge of that lizard. Lizards for which a fate could not be estimated were considered alive. To test for differences in the survival function (shape of the curve) between treatments, a log-rank test was used (Pollock et al. 1989). A Z-test statistic was also used to compare the survival curves on the last day of the summer (August 15; Pollock et al. 1989).

Table 1.—Number of ants at the bait station upon arrival (n = 4 in each treatment) for the active and inactive seasons on the Chaparral Wildlife Management Area, summers 1999 and 2000. The treatments are designated as U-U (control), Mg-B (Moderately grazed, burned), Mg-U (Moderately grazed, unburned, Hg-B (Heavily grazed, burned), and Hg-U (Heavily grazed, unburned)

	Active Season		Inactive Season	
Treatment	$\bar{x}$	SE	$\bar{x}$	SE
U-U	8.3	4.5	5.3	1.6
Mg-B	5.8	2.4	28.6	3.9
Mg-U	1.8	1.3	5.2	2.1
Hg-B	9.6	3.0	16.6	3.0
Hg-U	2.8	1.2	4.8	1.4

Results

Ant Abundance and Activity

More harvester ants were found at the bait station in the burned pastures, but this effect varied by season and level of grazing (3-way interaction, $F_{1,36} = 5.00$, $P = 0.03$, Table 1.). In both seasons, more ants were found in the burned pastures than the unburned pastures; however, this was especially true in the inactive season. More ants were found in all sites in the inactive season. In the active season, similar numbers of ants were found in the burned sites. In the inactive season, more ants were found in the moderately grazed/burned site than the heavily grazed, burned site ($P < 0.01$). In both seasons, the number of ants at the bait station was similar in the unburned sites. More ants were found in the control than the unburned, grazed sites ($F_{1,75} = 7.66$, $P < 0.01$). Fewer ants were found in the control than the 4 grazed sites ($F_{1,75} = 15.72$, $P < 0.01$) and the heavily grazed, burned site ($F_{1,75} = 12.24$, $P < 0.01$, Figure 1).

Burning affected the number of ants that arrived at the bait stations in one minute, but this effect varied by season and year (3-way interaction, $F_{1,36} = 3.30$, $P = 0.07$, Table 2). More ants were found in the burned sites in both seasons and both years than the unburned sites ($P < 0.04$ for all comparisons), except for the active season of 1999 ($P = 0.47$). In both years, more ants were found in the inactive season than active season for burned sites ($P < 0.01$ for both comparisons). More ants were also found in the burned sites in 2000 when compared to 1999 for both seasons ($P < 0.01$ for both comparisons). Finally, the number of ants that visited the bait stations was similar in the unburned sites for both seasons and both years.

Grazing also affected the number of ants that arrived at the bait station, but this effect varied by season and year (3-way interaction, $F_{1,36} = 7.55$, $P < 0.01$, Table 1). In the

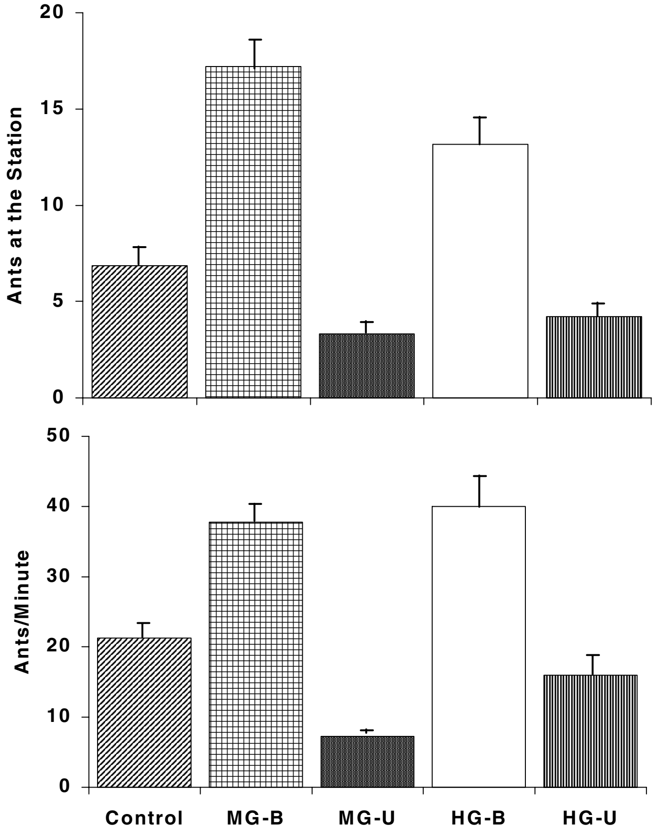

Figure 1.—Contrasts to the control for the number of ants at the ant bait stations upon arrival and the number of ants that visited the station within one minute at the Chaparral Wildlife Management Area, summer 1999-2000. The treatments are designated as Mg-B (moderately grazed, burned), Mg-U (Moderately grazed, unburned), Hg-B (Heavily grazed, burned), and Hg-U (Heavily grazed, unburned).

active season, the number of ants that visited the bait stations was similar in the moderately and heavily grazed sites. However, in the inactive season of 1999, more ants were found in the moderately grazed sites than the heavily grazed sites ($P = 0.05$); whereas in 2000 during the inactive season, more ants were found in the heavily grazed sites than the moderately grazed sites ($P < 0.01$). In both years, more ants were found in the inactive season than the active season for all sites ($P < 0.01$ for all comparisons) except for the heavily grazed sites in 1999 ($P = 0.51$). More ants were found in 2000 than 1999 for both seasons and levels of grazing ($P < 0.04$ for all comparisons).

More ants arrived at the station in the control than for the 4 grazed sites ($F_{1,75} = 3.52$, $P = 0.06$), but fewer than in the heavily grazed, burned site ($F_{1,75} = 5.86$, $P = 0.01$, Figure 1). More ants were found in the control than the unburned, grazed sites, but this difference was not significant ($F_{1,75} = 1.23$, $P = 0.27$). Though this comparison was not tested, fewer ants were found in the control than the burned, grazed sites (Figure 1).

Home Ranges

A total of 78 seasonal home ranges from 57 lizards were used in home range analyses. Total area used by horned lizards across both seasons ranged from 0.02 ha to 11.05 ha for 95 percent MCP (Table 3). The effect of burning on home range size interacted with season for 95 percent MCP ($F_{1,14} = 3.49$, $P = 0.08$). In the active season, home ranges in the burned sites (‘x ± SE = 1.14 ± 0.27 ha, $n = 18$) were smaller than those in the unburned sites (2.01 ± .06, $n = 19$), but were smaller and similar in size in both sites during the inactive season. All other interactions and main effects were not significant. Grazing did not have an effect on home range size ($P = 0.15$). Average (± SE) home range size for lizards in the control was 0.66 (± 0.22) in the active season and 0.80 (± 0.28) in the inactive season. None of the contrasts to the control were different.

Survival

Summer (15 Apr - 15 Aug) survival rates (*S*) ranged from 0.25 to 0.62. Grazing influenced survival rate of lizards ($P = 0.05$, Figure 2). Survival rates of lizards in the moderately grazed sites ($S = 0.60$) were higher than those in the heavily grazed sites ($S = 0.36$). Burning did not affect summer survival rates ($P = 0.19$, Figure 3). Because of a small sample size in the control, lizards from the control were not used in these analyses, but summer survival rate in the control was 1.00. However, 2 of 4 lizards died in the September - October period.

Discussion

The diet of the horned lizard consists primarily of ants (Burt 1928; Milne and Milne 1950; Pianka and Parker 1975; Whitford and Bryant 1979; Rissing 1981; Munger 1984*a;* Munger 1984*b*; Schmidt et al. 1989). Pianka and Parker (1975) found that 69 percent of the diet of Texas horned lizards was composed of harvester ants, with beetles composing the remainder. Numbers of harvester ants, therefore, could be one of the main components determining optimal habitat for a horned lizard. Previous studies on the effects of burning and grazing on ants showed that ants are not deleteriously affected by fire or grazing (McCoy and Kaiser 1990; Fox et al. 1996). The large increase in ant numbers for all ant indices in burned sites compared to unburned sites implies that on the CWMA, burning was beneficial to harvester ants. This conclusion was supported by the contrasts to the control. Ant numbers in the ungrazed, unburned control were intermediate to low values on the grazed, unburned sites, and high values on the grazed, burned sites.

The interpretation that burning had a positive effect on ant activity and distribution was complicated by interacting effects with grazing, season, and year. However, examination of these interactions indicated that, with regard to burning, the treatment effect varied

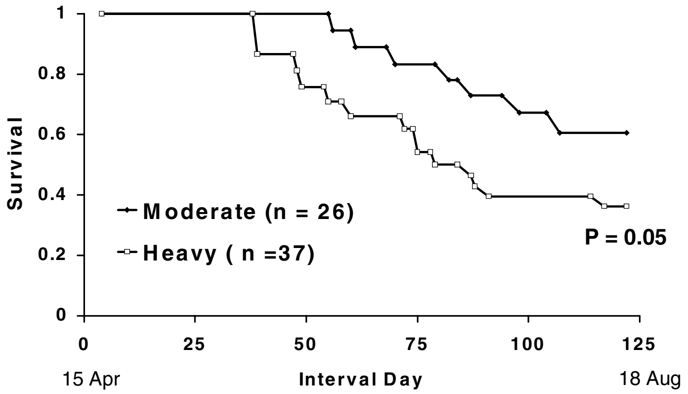

Figure 2.—Survival rates of Texas horned lizards in the moderately grazed (*S* = 0.60, 95% CI = 0.37-0.83) and heavily grazed sites (*S* = 0.36, 95% CI = 0.16-0.55) of the Chaparral Wildlife Management Area, summers 1998-2000.

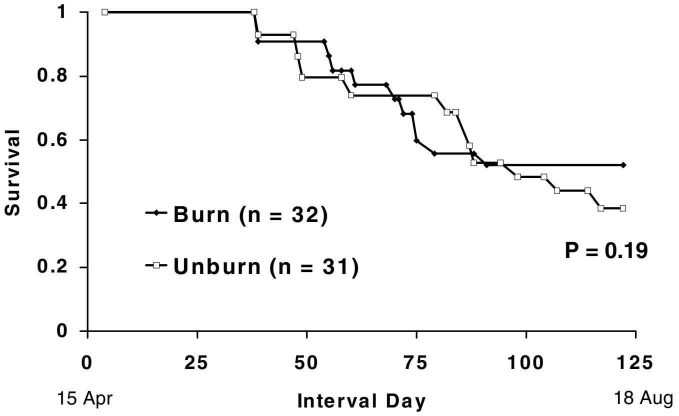

Figure 3.—Survival rates of Texas horned lizards in the burned (*S* = 0.52, 95% CI = 0.30-0.73) and unburned (*S* = 0.38, 95% CI = 0.16-0.60) sites of the Chaparral Wildlife Management Area, summers 1998-2000.

Table 2.—Number of ants that visited the bait stations within one minute (n = 4 in each treatment) for the active and inactive seasons on the Chaparral Wildlife Management Area, summers 1999 and 2000. The treatments are designated as U-U (control), Mg-B (Moderately grazed, burned), Mg-U (Moderately grazed, unburned, Hg-B (Heavily grazed, burned), and Hg-U (Heavily grazed, unburned)

	1999				2000			
	Active Season		Inactive Season		Active Season		Inactive Season	
Treatment	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
U-U	1.3	1.0	7.7	2.9	35.5	10.7	40.4	12.9
Mg-B	3.1	1.5	45.7	12.8	28.4	73.7	73.7	11.3
Mg-U	0.1	< 0.1	5.8	2.5	12.8	6.7	10.2	5.1
Hg-B	9.4	4.9	15.2	5.1	33.4	8.4	101.8	17.3
Hg-U	1.2	0.2	5.6	2.2	17.5	5.4	39.1	7.2

Table 3.—Home range sizes of Texas horned lizards (ha) using 95% Minimum Convex Polygon (MCP) for active and inactive seasons on the Chaparral Wildlife Management Area, summer 1998-2000. The treatments are designated as U-U (control), Mg-B (Moderately grazed, burned), Mg-U (Moderately grazed, unburned), Hg-B (Heavily grazed, burned), Hg-U (Heavily grazed, unburned)

	Active Season					Inactive Season				
Treatment	n	$\bar{x}$	SE	minimum	maximum	n	$\bar{x}$	SE	minimum	maximum
U-U	6	0.66	0.22	0.06	1.28	4	0.80	0.28	0.22	2.11
Mg-B	8	1.04	0.50	0.04	4.26	10	0.32	0.10	0.02	0.96
Mg-U	9	1.33	0.20	0.25	2.11	5	0.23	0.05	0.10	0.35
Hg-B	10	1.22	0.31	0.03	3.13	6	0.80	0.28	0.22	2.11
Hg-U	10	2.62	1.02	0.06	11.05	10	0.49	0.02	0.04	1.93

only in magnitude by season and year. The direction of the burning effect did not vary with time. Foraging activity in Chihuahuan desert harvester ants varied with season and year, with greater numbers of foragers in July and August than May and June (Whitford and Ettershank 1975). The greater number of harvester ants on the CWMA in the inactive season could be a result of increased foraging effort by harvester ants caused by greater seed availability in the inactive season, coupled with an increase in the number of foragers from reproductive efforts in early summer (Whitford and Ettershank 1975). Harvester ants are also thermophilic (Holldobler and Wilson 1990), so it is possible that the hotter temperatures associated with the inactive season enable harvester ants to forage more.

Increased seed availability and reproduction can also explain year effects. Whitford and Ettershank (1975) stated that harvester ant activity was regulated by seed availability and colony satiation. Perhaps 2000 was an exceptionally good year for seed production or reproduction resulting in more foraging ants. Increased ant activity in 2000 also could be a result of depleted resources from 1999. If granaries were depleted in 1999 due to a bad seed year, foraging effort would increase in 2000 to attempt to replenish the granaries. Low seed production in 1999 would also result in reduced activity in that year (Whitford and Ettershank 1975), thereby reducing the number of foraging ants at the bait stations. Unfortunately, we do not data on seed production to support these speculations.

Previous studies concluded that livestock grazing did not affect ant numbers in desert ecosystems (Heske and Campbell 1991; McClaran and Van Devender 1995:165). Grazing had a variable effect on harvester ant numbers, and appeared beneficial to ants when coupled with burning (Figure 1.1). As with burning, effects were stronger in the inactive season. The effect of level of grazing on ant activity interacted with year and specific ant index. For example, in the inactive seasons, more ants arrived at the station in moderately grazed sites in 1999, but in heavily grazed sites in 2000. However, more bait stations were consistently visited by ants in heavily grazed sites. In addition, higher ant indices were seen on the heavily grazed/burned site in all contrasts with the control. More bare ground in the heavily grazed site could be responsible for the increase in ants in heavily grazed sites in 2000, because harvester ants prefer areas of sparse vegetation (Holldobler and Wilson 1990). Therefore, it appears that heavy grazing was more beneficial to harvester ants, especially when coupled with burning.

Fire and grazing can improve conditions for harvester ants in several ways. Fire can increase forb and grass cover (McClaran and Van Devender 1995:134; Ruthven et al. 2000) and available bare ground, and decrease litter accumulation. Grazing can also increase forb abundance and decrease litter accumulation (Kelting 1954). Because harvester ants are granivores, most activity occurs in areas with interspersed bare ground and herbaceous vegetation. Forb and grass seeds provide the ants with food, and the sparse vegetation facilitates foraging (Holldobler and Wilson 1990). Finally, DeMers (1993) noted that harvester ant queens prefer to start a new mound in open areas with little vegetation. An increase in prey abundance associated with burning could result in a subsequent increase in Texas horned lizard density or a decrease in ranging behavior.

Home range size is inversely proportional to the distribution and abundance of resources for many species, including several lizards (Mares et al. 1976; Litvaitis et al, 1986; Boutin 1990; Lacher and Mares 1996). Little is known about the size of home ranges for the Texas horned lizard, although information does exist in closely related species (Lowe and Stebbins 1954; Baharav 1975; Turner and Medica 1982). Munger (1984*c*) reported home range sizes of Texas horned lizards in Arizona as averaging 1.35 ha for females (n = 13) and 2.40 ha for males (n = 10). Home ranges in my study were considerably larger than those previously reported by Fair and Henke (1999), who estimated home range size of Texas horned lizards in southern Texas to be between 0.02 to 1.47 ha (n = 16). However, their home range estimates were only the sum of weekly estimates of several months.

We propose that the smaller home ranges of horned lizards in burned pastures resulted from burning improving the habitat of horned lizards to the degree that ecological requirements (i.e., food, cover) were found in a smaller area. Because grazing did not affect home range size, it is possible that grazing at the intensities studied is neither beneficial nor harmful to the habitat of horned lizards. The mechanism by which burning and moderate grazing may improve horned lizard resource distribution, and thus reduce home range size, is by creating open habitats interspersed with vegetation cover. Whiting et al. (1993) found that Texas horned lizards selected disturbed habitats over undisturbed habitats. They suggested that prey abundance and suitable open habitats were major factors related to the spatial occurrence of Texas horned lizards in Texas. Disturbances that create an open, sparsely vegetated habitat appear to benefit horned lizards in several ways. Open areas facilitate movement by this dorso-ventrally flattened species (Whiting et al. 1993). Fair and Henke (1998) also found that Texas horned lizards selected recently burned areas compared to areas with large litter accumulation because of increased mobility. Pianka (1966) found that horned lizards preferred open areas to sit and wait for their prey, thus increasing foraging efficiency. Open habitats also aid in thermoregulation by allowing horned lizards exposure to direct solar radiation (Heath 1965). Finally, horned lizards may select open habitats due to an increase in food abundance, specifically of harvester ants.

Smaller home ranges in the inactive season were expected based on our observations of horned lizard activity. Fair and Henke (1999) also found that home ranges decreased in size as the summer progressed until hibernation. Several reasons could explain seasonal differences in activity and ranging behavior of horned lizards. First, increased mobility of horned lizards during late spring and early summer could be due to mate-searching and nest-building activities. Horned lizards on the CWMA typically emerge from hibernation in early March or April and become highly mobile, often moving > 100 m/day to reproduce, build nests, and lay eggs. Second, as the summer progressed, the temperatures rose to points that could be lethal to horned lizards (Forrester et al. 1998); therefore, horned lizard movements were likely constrained by temperature in the inactive season. Third, the increase in harvester ant abundance and activity in the inactive season may have enabled lizards to move shorter distances to find food, thereby reducing the ranging behavior of the lizards.

Estimates on survival rates of horned lizards are contentious due to the large number of censored lizards. Munger (1986) found that Texas horned lizards in southeastern Arizona had annual survival rates between 35.0 and 86.0 percent, whereas Fair and Henke (1999) estimated 8-month survival rates (Mar-Oct) in southern Texas to be lower (8.9 -54.0 percent). However, Pianka and Parker (1975) suggested adult Texas horned lizards have comparatively high survival rates. Our estimates (25 - 62 percent) fall between those of Munger (1986) and Fair and Henke (1999). Lower survival rates in the heavily grazed sites suggested that heavier levels of grazing increase the vulnerability of horned lizards to mortality and may counteract increased prey abundance. Burning did not affect summer survival rates of horned lizards, though survival rates were higher in burned than unburned sites during the inactive season ($P = 0.03$). Higher survival rates in moderately grazed sites and burned sites could be due to better juxtaposition of food and cover, as supported by the home range and ant data.

The smaller home ranges, increased survival rates (at least in the inactive season), and greater prey abundance in burned pastures suggested a positive effect of fire on the ecology of Texas horned lizards. The effect of grazing was more complex. Survival was decreased in heavily grazed pastures, but range size did not differ among grazing level. Also, ant activity was generally higher in the heavily grazed pastures, especially when coupled with burning. Our comparisons to an ungrazed, unburned control were limited by a small sample in the control pasture. However, measures of lizard resources (e.g., vegetation, ants) and performance (e.g., range size, survival) were generally comparable between control and treated pastures. An alternate-year burning regime and stocking rates of livestock such as those implemented by CWMA created suitable habitat for Texas horned lizards in southern Texas.

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Fire and Bird Communities in the South

James G. Dickson[1]

Abstract.—Fire has long been a natural and anthropogenic force shaping southern forests and their fauna. Some species are attracted to recent burns. There is little direct mortality of adult birds by fire, but growing season fires may consume some nests. Fire affects bird communities mainly through effects on vegetation. Fires effective enough to limit understory hardwood development would reduce habitat suitability for associated birds, but would promote species associated with grass-forb vegetation. In the long term, fire would disfavor species associated with deciduous foliage and favor species associated with pine canopies.

Introduction

For thousands of years fire has been a natural and anthropogenic force shaping southern forests and their fauna. Southern forests and the animals that inhabit them evolved with fire, and fire has helped shape southern systems. Lightning-set fires have been a recurring force (Heyward 1939), natives in the region used fire to manipulate vegetation and drive game for harvest, and European settlers used fire for the same purposes and to clear new land for their crops.

Pine savannahs of the southern Coastal Plain with their grassy understories were fire adapted. Longleaf pine and red-cockaded woodpecker are fire adapted species, and fire played a role in perpetuating the upland oak forests. Natural fires periodically still affect southern forests and prescribed burning continues to be used for a variety of purposes, including wildlife management. Currently, prescribed fire is used extensively to maintain pine ecosystems, and particularly to manage for the red-cockaded woodpecker.

In this publication I treat the effects of fire on birds in the South. There are some recent data on fire and birds in the South. And inferences can be drawn from data from other regions, general information about effects of fire on vegetation, and bird-habitat relationships (Dickson 1981).

Variability

Fires and their effects on systems are extremely variable. Every fire and its effects are different; and usually there is considerable variation within each fire. Fires and their effects involve habitat conditions before burning and suitability for a wide variety of different communities or species; the intensity, periodicity, and seasonality of fires; landscape context and unburned areas, and numerous interactions. For example, a low intensity winter fire with high relative humidity, high fuel moisture, carried by a low wind over a moist forest floor has little long-term impact. Conversely, intense conflagrations have long term effects on systems and their components. I approach this treatment by describing how fire may affect forests, and how that may affect bird species and communities.

Short Term Effects

Obviously, fire has the potential to kill animals; and generally, animals with low mobility are the most vulnerable. There is documentation of some mammals perishing in fires (Bendell 1974), and probably some reptile and amphibian mortality. But most animals can escape fire by fleeing, finding unburned shelter, or burrowing. There probably is little direct mortality of birds from fire, although growing season fires probably consume some nests located on or near the ground. For example, Doerr (et al. 1970) observed that no ruffed grouse (see Tables 1 and 2 for scientific names of animals and plants, respectively) were directly killed by a fire but grouse nests apparently were consumed. Direct mortality of animals from fire on a broad population or landscape scale is negligible.

Some birds actually are attracted to the heat and smoke of fires, or to the burns shortly thereafter. A list of 77 species of North American birds that were attracted to fire and smoke or observed on recent burns was compiled by Komarek (1969). Wild turkeys and mourning doves are attracted to new burns in search of exposed insects and seed. Raptors, such as red-tailed hawks, kestrels, other hawks, and owls have been observed attracted to burns in search of prey (Stoddard 1963, Landers 1987). Other species attracted to recent burns include purple martins, American robins, eastern bluebirds, sparrows, pine warblers, common flickers, and other woodpeckers.

Habitat

Generally fire consumes forest floor litter and sets back plant succession; usually reducing smaller hardwoods in favor of thicker-barked pines and herbaceous vegetation. Small top-killed hardwoods usually resprout after infrequent and/or cool season fires; but conversely, small hardwoods can be severely reduced or eliminated by repeated, and especially growing-season burns (Lewis and Harshbarger 1976). Hardwood trees and shrubs impede more light to the forest floor than pines; therefore, burning severe enough to reduce hardwoods would favor vegetation near the ground, especially

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[1] School of Forestry, Louisiana Tech University, Ruston, LA 71272

grasses and forbs. In upland oak-hickory stands burning favors mid-successional oaks which resprout, over the succession of more tolerant hardwoods.

Fire affects hardwoods and their suitability for birds in other ways. Fire wounds on hardwood trees provide entrance for decay, which over time may become excavated into cavities by woodpeckers, and used by primary and subsequently secondary cavity nesters. Severe fire may kill trees and create snags that are used by a variety of cavity-using wildlife. But conversely, standing dead snags used by cavity nesters may be consumed by fire.

Bird Communities

In the short term there is some site fidelity of birds; many remain on the site in spite of burn-induced habitat changes (Bendell 1974). In south Florida there was little immediate effect of burning of a slash pine stand on the bird community (Emlen 1970). Emlen concluded that bird attachments to home ranges and familiar foraging areas transcended species habitat selection immediately following a fire.

Burning usually does affect bird communities through habitat changes for several years postburn (e.g., Engstrom et al. 1984, White et al. 1999). Bendell (1974) summarized a number of studies of effects of burning on birds. He concluded that overall, the number of species of birds increased after burns. The number of species that fed on or near the ground increased the most, as might be expected with the decrease of trees and the proliferation of grasses, herbs, and shrubs after burning. The number of tree canopy species decreased the most.

Some specific examples of the influence of fire on bird communities illustrate. In a pine stand in northern Florida that had been burned regularly, a number of species dwindled over time with fire exclusion as dense hardwood sapling stand developed (Engstrom et al. 1984). With 15 years of fire exclusion open habitat species disappeared quickly, shrub associated species increased then dwindled, and mesic forest species invaded.

In a study of prescribed burned mature pine stands the Georgia Piedmont avian species richness was similar between burned and unburned stands, but about 3 times as many species preferred the burned stands (White et al. 1999). Burned stands were much preferred over unburned for nesting. But productivity was very low on burned sites, with predation implicated as the major cause of nest failure.

A somewhat similar pattern in the bird community was observed with hardwood removal (WSI) and burning in the Ouachita Mountains of Arkansas (Wilson et al. 1995). Species richness was similar among treatments. However, bird densities were highest the second year after WSI and burning, and lowest in unburned controls. Densities of ground/shrub-foraging and shrub-nesting species increased the most following WSI and fire. Ground-nesting species were more abundant in untreated stands.

Bird populations were tracked for 15 years after a wildfire in a coniferous forest in California (Bock et al. 1978). After 8 years, the bird community was slightly richer and more diverse in burned than unburned forest. Open ground and brush species predominated on the burned areas, whereas coniferous foliage feeders predominated on the unburned area. Eight to 15 years post burn ground foragers and cavity nesters declined on the burned area. After 15 years the burned area had become brushy and most fire-killed trees had fallen. Bird density and diversity were similar on the burned and unburned areas.

A wildfire in ponderosa pine forests in Arizona drastically affected the bird community (Lowe et al. 1978). The number of birds increased the first year after the burn, but decreased substantially from 5 to 20 years. The abundance of timber gleaning, flycatching, aerial flycatchers, tree foliage searching, ground and brush foraging birds, and timber drilling birds changed over time in response to the burn.

From 2 to 4 years after a Minnesota wildfire 54 species decreased, 17 species increased, and 42 species remained relatively unchanged (Niemi 1978). Waterbirds, flycatchers, swallows, vireos, and warblers decreased the most; while woodpeckers increased the most.

Fire also affects birds other than during the breeding season. For example, during winter in a northwestern coniferous forest that was burned by a wildfire, the number of bird species was similar between burned and unburned stands, but composition was distinctly different (Kreisel and Stein 1999). Trunk and bark foraging species, mostly woodpecker, were twice as abundant in burned stands than unburned, but declined in abundance in subsequent years after the burn.

Bird Species in Southern Forests

A few species, such as the red-cockaded woodpecker, brown-headed nuthatch, and pine warbler are associated closely with mature pine trees (Johnston and Odum 1956). Fires severe enough to preclude hardwoods and favor pines would favor these species. Birds were contrasted in mature pine stands burned regularly, in part, for the red-cockaded woodpecker, versus unburned pine-hardwood stands the Georgia Piedmont (White et al. 1999) and in the Ouachita

Mountains of Arkansas (Wilson et al. 1995). In both areas there were no red-cockaded woodpeckers in the unburned stands with hardwoods, and significantly fewer brown-headed nuthatches and pine warblers.

But most bird species in pine stands are associated with non-pine vegetation. For example, in pine stands in Georgia the hardwood understory largely determined the nature of the bird community (Johnston and Odum 1956). And in eastern Texas middle-aged pine stands with no hardwoods were virtually devoid of birds (Dickson and Segelquist 1979). With fire, the composition and status of bird communities would be largely determined by habitat responses to burning (Dickson et al. 1993).

Typically, frequent, intense, or growing season fire reduces small hardwoods and shrubs, and precludes sizable hardwoods from developing. The presence and abundance of birds associated with hardwood canopy, such as the yellow-billed cuckoo, great crested flycatcher, tufted titmouse, red-eyed vireo, black-and-white warbler, and worm-eating warbler would depend on the extent of hardwood canopy, that could be affected by the fire regime of an area. In the Ouachitas, black-and-white warblers were more abundant in stands with hardwoods than in stands where hardwoods had been controlled (Wilson et al. 1995).

Several species, such as the Acadian flycatcher, Carolina wren, hooded warbler, Kentucky warbler, and northern cardinal are associated with shrub-level and hardwood midstory vegetation (Dickson and Noble 1978). The distribution and relative abundance of these species could depend on the extent of appropriate habitat, which could be determined by fire (Dickson 1981). As a hardwood sapling subcanopy developed with fire exclusion in a pine stand in northern Florida, species associated with mesic woods i.e., yellow-cuckoo, wood thrush, red-eyed vireo, and hooded warbler, increased (Engstrom et al. 1984). In the Georgia Piedmont, the northern cardinal was more abundant on unburned sites, but the Acadian flycatcher and Carolina wren were more abundant on burned sites (White et al. 1999). In the Ouachitas, both the cardinal and Carolina wren thrived in stands where hardwoods had been killed but had not been burned (Wilson et al. 1995).

The reduction of hardwoods through fire would promote the development of fire-maintained pine savannahs with associated grass-forb and low shrub vegetation, with abundant seeds and fruit (Lewis and Harshbarger 1976). This situation would benefit species associated with these conditions, such as the prairie warbler, common yellowthroat, yellow-breasted chat, field sparrow, Bachman's sparrow, grasshopper sparrow, eastern towhee, and indigo bunting (Johnston and Odum 1956, Meyers and Johnson 1978, Dickson and Segelquist 1979, White et al. 1999). This relationship is supported by fire exclusion data from the Coastal Plain. Stands developed from a pine savannah, to shrub, to hardwood subcanopies (Engstrom et al. 1984). Bachman's sparrows were soon eliminated from the stand. After a few years yellow-breasted chats, common yellowthroats, and indigo buntings were virtually gone. Eastern towhees and northern cardinals responded positively to developing shrubs, but dwindled as hardwood trees dominated. In the Ouachitas, chipping sparrows, indigo buntings, and Bachman's sparrows were more common on WSI/burn plots, and prairie warblers numbers increased with years post-burn as hardwood shrubs developed (Wilson et al. 1995).

Grass-forb vegetation would also provide important brood habitat for southern gallinaceous species: northern bobwhites and wild turkeys. In the Ouachitas, both bobwhites and turkeys were detected regularly on the WSI / burn plots (Wilson et al. 1995).

As noted, there usually is much variation in fire effects on vegetation due to differences in local factors such as fuel loading, size, and moisture content; as well as local topography, soils, and daily weather. Fires which result in a variable habitat with some unburned patches would probably enhance bird abundance and diversity, and generally be positive for birds. For example, Roth (1976) successfully predicted bird species diversity in brushlands with a vegetation heterogeneity index. In the South (Piedmont, White et al. 1999; Ouachitas, Wilson et al. 1995), bird densities were higher in burned treatments than in controls, probably due to the patchy habitat resulting from fire. The eastern wood pewee, an edge species, responded positively to fire in the Ouachitas (Wilson et al. 1995). Other typical edge species of southern forests which may benefit from habitat variability resulting from fire would be: the great crested flycatcher, and Carolina chickadee (Strelke and Dickson 1980).

Fruiting

During fall and winter, fruits and seeds are a main diet item of birds (Martin et al. 1951). Burning affects fruit production and availability. Burning exposes seeds to foraging birds in forest floor litter. For example, in the Georgia Piedmont there were significantly more herbaceous plant and legume seeds on prescribed burned plots than on unburned plots the first growing season after burning (Cushwa et al. 1966). And in the Georgia Coastal Plain there was significantly less seed in subsequent years after burning (Buckner and Landers 1979).

Short and long term effects of fire on fruiting are more variable. In eastern Texas the first year after fire, the number of plants with fruit was reduced by 72% (Lay 1956). Fruiting yaupon, holly, sweetleaf, and viburnum were reduced. Fruiting dogwoods and American beautyberry increased.

After a couple of years post-fire, herbaceous vegetation is gradually replaced by hardwood sprouts and shrubs. Also, fruiting of shrubs recovering from fire and those benefitting from reduction of vegetative competition from fire increase. This fruit production benefits a number of fruit consuming species, such as the wild turkey and northern bobwhite (Landers 1987).

Arthropods

Also, although our knowledge of fire effects is limited, arthropod populations and their interactions with vertebrates certainly are affected by fire (Landers 1987). Some species of insects are attracted to the flame and smoke of fire, and reproduce in fire conditions (see Lyon et al. 1978). Evans (1972) noted that Platypezid and Empidid smoke flies, and some Cerambycid and Buprestid beetles appear to be attracted to smoke; and Melanophila oviposit in newly burned wood.

Effects of fire on arthropods on the forest floor seems quite variable. It is generally conceded that invertebrate soil fauna is reduced by fire; whereas surface insects are less vulnerable to fire (Lyon et al. 1978). It has been documented that parasites of wild turkey and northern bobwhite are reduced in the short term by burning (Dickson 1981).

Precautions

Red-cockaded woodpeckers peck cavity trees to produce a resin flow around the cavity and down the tree bole. Fire may ignite the resin up the tree bole of cavity trees and may even burn out and gut the nest cavity(Conner and Locke 1979). Therefore, surface fuel around cavity trees may need to be raked away before burning.

Streamside zones (SZ) are strips of mature hardwoods and pines along intermittent and permanent streams traversing upland stands, often pine. Retention of streamside zones (SZ) are important concessions for wildlife in southern forests. These zones serve as travel corridors for some species and provide important mature habitat for birds (Dickson et al. 1995). These moist hardwood-dominated sites usually will not carry a fire. But with arid conditions and low fuel moisture fire can a problem; and SZ should be protected from fire.

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Table 1.—Scientific names of animals

Red-tailed hawk	*Buteo jamaicensis*
American kestrel	*Falco sparverius*
Wild turkey	*Meleagris gallopavo*
Ruffed grouse	*Bonasa umbellus*
Northern bobwhite	*Colinus virginianus*
Mourning dove	*Zenaida macroura*
Yellow-billed Cuckoo	*Coccyzus americanus*
Common flicker	*Colaptes auratus*
Red-cockaded woodpecker	*Picoides borealis*
Great Crested flycatcher	*Myiarchus crinitus*
Acadian flycatcher	*Empidonax virescens*
Eastern wood-Pewee	*Contopus virens*
Purple martin	*Progne subis*
Carolina chickadee	*Parus carolinensis*
Tufted titmouse	*Parus bicolor*
Brown-headed nuthatch	*Sitta pusilla*
Carolina wren	*Thryothorus ludovicianus*
Wood thrush	*Hylocichla Mustelina*
American robin	*Turdus migratorius*
Eastern bluebird	*Sialia sialis*
Red-eyed vireo	*Vireo olivaceus*
Black-and-White warbler	*Mniotilta varia*
Worm-eating warbler	*Helmitheros vermivorus*
Pine warblers	*Dendroica pinus*
Prairie warblers	*Dendroica discolor*
Common yellowthroat	*Geothlypis trichas*
Yellow-breasted chat	*Icteria virens*
Kentucky warbler	*Oporornis formosus*
Hooded warbler	*Wilsonia citrina*
Northern cardinal	*Cardinalis cardinalis*
Indigo bunting	*Passerina cyanea*
Eastern towhee	*Pipilo erythrophthalmus*
Grasshopper sparrow	*Ammodramus savannarum*
Bachman's sparrow	*Aimophila aestivalis*
Chipping sparrow	*Spizella passerina*
Field sparrow	*Spizella pusilla*

Table 2.—Scientific names of plants

Longleaf pine	*Pinus palustris*
Slash pine	*Pinus elliottii*
Ponderosa pine	*Pinus ponderosa*
Holly	*Ilex opaca*
Yaupon	*Ilex vomitoria*
Dogwood	*Cornus florida*
Sweatleaf	*Symplocos tinctoria*
American beautyberry	*Callicarpa americana*

Burning for Birds: Concepts and Applications

R. Todd Engstrom[1] *and David J. Brownlie*[2]

Abstract.—Prescribed fire is being used extensively for habitat management of non-game birds, although the area burned today is small relative to the amount of land that burned historically. Results of a non-scientific questionnaire of public and private land managers in the eastern U.S. revealed prescribed fire is being used to provide winter, breeding season, and migration habitat for at least 57 species of birds in 29 states. Increasingly sophisticated application of fire will be necessary to manage habitat for diverse bird species and other organisms. Contemporary training courses on the use of prescribed fire typically divide ecological effects of fire on animals into two categories: direct effects alter the animals' physical condition and indirect effects are mainly associated with changes in habitat. The envirogram, a conceptual tool developed by Andrewartha and Birch (1984) within their theory of the environment, can be used to diagram multiple direct and indirect effects of fire on bird species. Variables of prescribed fire (i.e., ignition pattern, season, frequency, etc.) can be matched within the envirogram to achieve the desired management objectives for individual species. We used a modified envirogram to examine how prescribed fire is used in intensive management for the northern bobwhite (*Colinus virginianus*) within the Red Hills of north Florida and south Georgia. A coarse-grained approach to management involving multiple species or an ecosystem may be more efficient and sustainable than emphasis on a single species.

Introduction

Extensive use of fire to manipulate vegetation by Indians disappeared from eastern North America centuries ago, and natural fires (e.g., lightning-started) have been reduced in the eastern United States by habitat fragmentation, fire suppression, and alteration of vegetation composition. Wildfires can be widespread during times of prolonged drought—nearly one-half million acres were burned in 2282 wildfires from 1 June to 22 July 1998 in Florida (Lewis 1998)—but these wildfires may behave very differently from natural fires because of long periods of fire suppression. It is probably not a coincidence that both the use of fire as a management tool (Pyne 1982) and populations of bird species that require habitat that is in early stages of vegetation succession (Askins 1999) declined in the 20th century. Current uses of prescribed fire can only be considered a limited stabilization or reversal of that trend.

Fire plays a critical role in ecosystem management as an ecological process in many vegetation types in the United States (Smith 2000). Fire has been long been recognized as an essential disturbance in some ecosystems, such as longleaf pine (*Pinus palustris*) woodlands, which require a relatively high fire frequency to suppress competing hardwoods and maintain an open forest structure that fosters pine regeneration. The endangered red-cockaded woodpecker (*Picoides borealis*) depends on pine forest structure and composition that are shaped by frequent fire. In other ecosystems, such as oak woodlands, savannahs, and barrens in the central hardwoods region (Anderson and others 1999; Yaussy 2000), acceptance of the role of fire continues to grow. The finer points of prescribed fire will play an increasingly important role in the success of natural resource management as wildlands are reduced and multiple management objectives increase.

Prescribed fire is pervasive in the growing emphasis of modeling vegetation manipulations on natural disturbances within the context of ecosystem management (White and others 1999; Engstrom and others 1999). According to a recent review, ecosystem management is "…management driven by goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions, and processes necessary to sustain ecosystem composition, structure, and function" (Christensen and others 1996). In this definition, sustainability of resources is a precondition of management for production. Maintenance of viable populations of animals (e.g., game or endangered species) is one of many natural resource goals that must be balanced within the ecosystem management framework by a broader "ecosystem composition, structure, and function." From this perspective, understanding the ecological relations of a single species is a critical element of ecosystem management within the constraints of ecosystem sustainability.

In this paper we (1) summarize the results of a survey to better understand the extent and nature of prescribed fire applications to non-game bird management; (2) examine the ecological basis for use of prescribed fire to manage bird populations; and (3) examine application of prescribed fire within the context of management for a single-species, the northern bobwhite (*Colinus virginianus*) within the Red Hills.

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[1] Tall Timbers Research Station, Tallahassee, FL 32312

[2] U.S. Fish and Wildlife Service, Tallahassee, FL 32312

Use of Prescribed Fire for Bird Management un the Eastern U.S.: Survey Results

In the statement of purpose of this symposium, it was stated "...with the exception of a few high-profile threatened or endangered species, the use of prescribed burning solely for nongame species or communities has been a rare occurrence." The dependence of some threatened and endangered bird species (e.g., red-cockaded woodpecker) on fire-maintained habitats is well-established (James and others 1997). To obtain a baseline evaluation of the current uses of prescribed fire in relation to populations and communities of birds in general, we surveyed 97 organizations that we thought to be prescribed fire practitioners. We distributed our survey electronically to contacts in the U.S. Fish and Wildlife Service, USDA Forest Service, National Park Service, state agencies (i.e., Forestry/Fire Protection, Wildlife, and Natural Heritage/Endangered Species programs), and The Nature Conservancy in 29 eastern states. Fifty-eight of a total of 60 responses stated that they burned property they manage, and, of these, 42 (72%) of the respondents stated that they burned specifically to provide habitat for birds (Brownlie and Engstrom 2001).

Fifty-seven bird species in 22 families (77% were nongame species) were identified by respondents as species that benefit from land management with prescribed fire. (See Brownlie and Engstrom 2001 for a complete list of species). Predictably, grassland birds, such as the emberizid sparrows, were well-represented and comprised 23% of the total; prescribed fire was also used for management of gamebirds, such as quail, dove, and ducks. Although our assessment of birds that are managed with fire is general, it indicates that prescribed fire is being used extensively for game and nongame birds, not just threatened and endangered species and that prescribed fire for management of avian habitat is being applied in a wide variety of vegetation types throughout the eastern United States. The diversity of habitats that are being burned for a diversity of bird species implies that many techniques of prescribed fire are being used.

Concepts

Ecology provides the scientific foundation of ecosystem management. One of the concepts critical to an ecological understanding of the relationship between birds and fire is plant succession. Christensen (1988) developed a model of succession that involved four phases: establishment, thinning, transition, and steady state. In establishment, seedlings grow rapidly. In the second phase, plants thin out during a period of intense inter- and intra-specific competition. The third phase involves gaps created in the canopy and initiation of small-scale recruitment. In the last phase, the plant community reaches a steady state in which establishment, thinning, and gap formation occur in a mosaic of patches. The scale and type of disturbances influence the size and nature of plant response to the patches. It is important to note that disturbance is implicit in natural plant community development. The stability of the "steady-state" is dependent on a variety of factors. Alteration of the disturbance regime or plant community composition (e.g., invasion of exotics) can disrupt community equilibrium and potentially shift the community into a new steady state (Wiens 1997).

Making theory relevant to managers is a substantial challenge to ecologists and requires a constant search for techniques that provide an interface between the two disciplines: science and management (Rodgers 1997). We use a simple conceptual technique, the envirogram (Andrewartha and Birch 1984), as such an interface to review some of the concepts that are essential for an ecological understanding of the relationship between fire and birds that is necessary for ecosystem management.

Effects of fire on animals are frequently classified as relatively short-term and direct (mortality or movement) or long-term and indirect via plant succession and alteration of vegetation structure (Smith 2000). This simple dichotomy is expanded within the envirogram (Figure 1; Andrewartha and Birch 1984). The envirogram is composed of three parts: the focal organism, the centrum (direct effects), and the web (indirect effects). The ability of the focal organism to survive and reproduce is influenced by four components of the centrum—resources, mates, predators, and malentities. The direct effects within the centrum are affected by environmental influences within the web that have increasingly less direct influence on the organism of interest. Fire plays multiple positive and negative roles within the centrum and the web for different organisms. For birds, fire can be within the centrum as a malentity, because it can kill an organism. At the same time, fire can be used to improve the structure of the habitat over a longer period.

In our envirogram, we match characteristics of prescribed fire with aspects of the environment that affect an animals' ability to survive and reproduce (Figure 1). Fire can be prescribed to increase or decrease its effect as a malentity by manipulating method of application and season. Season and severity can be altered to influence availability of food and nesting substrate. Area of burned acres on a landscape scale will play an important role in creating or eliminating habitat that will affect characteristics of the metapopulation of the species of interest. Patchiness of a burn locally may change the suitability of an area by altering shelter from predators and proximate sources of food.

James and others (1997) used the envirogram as a graphical technique to summarize the life history of the red-cockaded woodpecker. In their envirogram, fire

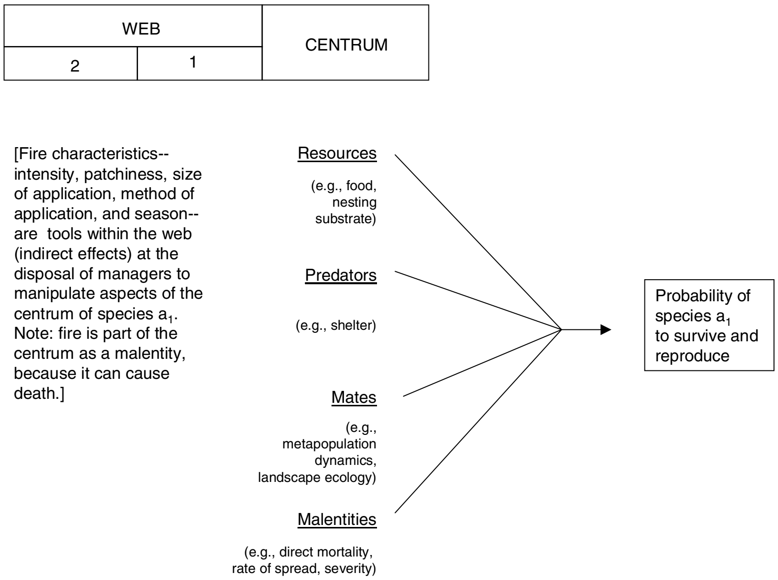

Figure 1.—Modification of a generalized Andrewartha and Birch (1984) envirogram to represent the various aspects of prescribed fire that indirectly and directly affect the ability of a bird species to survive and reproduce. Given the characteristics of an avian population of management interest prior to use of prescribed fire, a manager can match ecological characteristics of a bird species with the prescribed fire objectives.

played an indirect role in the amount of foraging habitat (a resource) through the effects on the abundance and distribution of hardwoods that compete with pines. They also hypothesized that fire might play a role in nutrient availability and population levels of cavity competitors. Their research indicated that frequency and season of fire were critical indirect influences on measures of productivity of the woodpecker, although they could not identify the exact causes of the relationship. We used the envirogram for another bird species that is managed intensively, the northern bobwhite.

Birds and Prescribed Fire in the Red Hills Landscape

The Red Hills physiographic region between Tallahassee, Florida, and Thomasville, Georgia, includes approximately 100 privately owned parcels in 310,000 acres of land that are managed for hunting (C. Ambrose pers. comm.). White-tailed deer (*Odocoileus virginianus*), mourning dove (*Zenaida macroura*), and ducks, primarily wood duck (*Aix sponsa*) and ring-necked duck (*Athya collaris*), are hunted, but northern bobwhite (*Colinus virginianus*) is the raison d'être for the hunting estates. Management for populations of bobwhite in the Red Hills plays a notable chapter in the development of prescribed fire as a management tool through the contributions of Herbert L. Stoddard, Ed and Roy Komarek, and Tall Timbers Research Station (Pyne 1982; Johnson and Hale, this volume).

Approximately 90 to 100 bird species are resident, breed, or overwinter throughout longleaf pine woodlands (Engstrom 1993). Depending on ground cover characteristics, approximately one third of the bird species, including the northern bobwhite, may nest and forage on or near the ground in longleaf pine woodlands (Engstrom 1993). These ground-nesting and ground-foraging species are presumably the species that are most directly affected by fire. Many of the bird species of the longleaf pine ecosystem are also common in old-field pinelands, which are structural analogues of longleaf pine woodlands (Engstrom et al. 1984), although a detailed comparison has never been made. Three species with affinities for open pine woodlands—the northern bobwhite, Bachman's sparrow (*Aimophila aestivalis*), and red-cockaded woodpecker—disappeared from an open oldfield pine woodland in northern Florida (NB66, a 22-acre study area on Tall Timbers Research Station) within 4 to 15 years after fire exclusion (Figure 2). This plot is now a closed canopy mixed hardwood-pine woodland with little ground-cover vegetation.

Based on evidence from existing remnant uncultivated tracts, pre-settlement upland vegetation on the hunting estates was likely an open-structured woodland dominated by longleaf pine with scattered species of other southern pines and fire-resistant hardwoods (e.g., *Quercus*, *Carya*) with a species-rich understory (Platt et al. 1988). Ground cover plant species richness in mesic longleaf pine woodlands at a 1000 m^2 scale is among the largest values reported in the temperate Western Hemisphere (Peet and Allard 1993). Some parts of the Red Hills, particularly in Florida, were converted to agriculture in the first half, and turned fallow in the latter half, of the 19th century (Paisley 1989). Agriculture eliminated much of the longleaf pine and some of the pyrogenic ground cover plants (e.g., wiregrass [*Aristida stricta*]) on many of these old-field sites. Loblolly (*P. taeda*) and shortleaf (*P. echinata*) pines frequently replaced longleaf as overstory dominants.

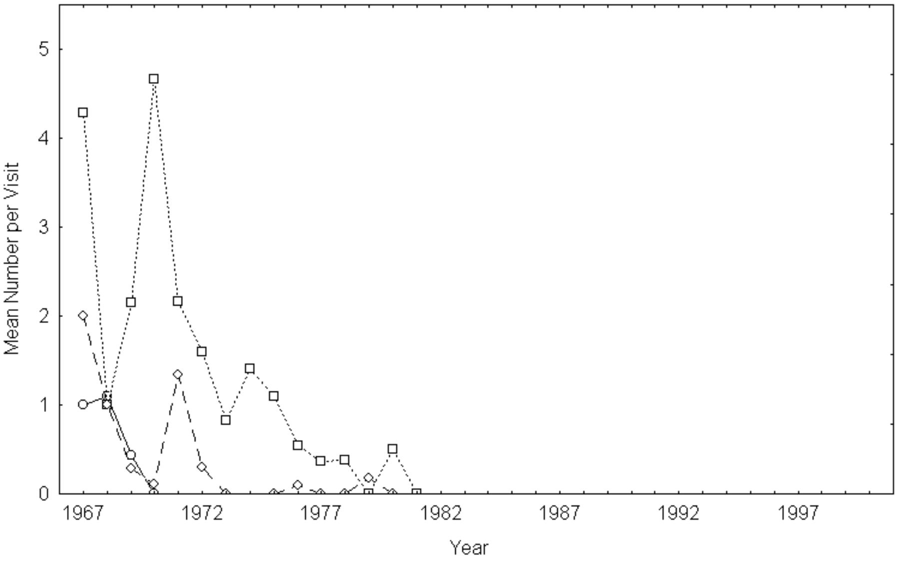

Figure 2.—Changes in abundance of northern bobwhite (dotted), red-cockaded woodpecker (dashed), and Bachman's sparrow (solid) in an oldfield pine woodland over a 15-year period in response to fire exclusion (adapted from Engstrom et al. 1984).

Typical historical fire behavior within longleaf pine woodlands is believed to have been frequent (1 to 3 year fire intervals) and low intensity that rarely killed mature trees (Myers 1990). These fires reduced aboveground herbaceous vegetation biomass to ash and killed or topkilled most hardwood stems. Grasses and forbs readily resprout after these low intensity fires. The season of natural fires, based on lightning-started fires, is strikingly different from the seasonal application of prescribed fire for management of northern bobwhite populations on the hunting estates (Figure 3). Prescribed fires on the hunting estates are typically initiated immediately after the end of quail hunting season in late-February , March, or early April, whereas lightning-started fires occurred most frequently in May or June (Komarek 1968; Brennan and others 1995). The choice of season for prescribed fire in the Red Hills is based on the desire to avoid burning during the hunting season, minimize loss of quail nests to fire, take advantage of dry vegetation and relatively cooler, drier weather, and prepare for timber management activities. Many of the prescribed fires in the Red Hills are small to intermediate scale (10 to 200 acres) although sometimes as much as 1000 acres may be burned on a single land ownership in a day. Landscape patterns of Red Hills prescribed fire is typically controlled by using roads or plowed lanes as firebreaks. Fires are mostly conducted during the daytime (especially in Florida) under low humidity, and light wind conditions, but nighttime fire is permitted in Georgia. These occasional nighttime fires often result in patchy low-severity burns. Flanking, backing, and head fires are used depending on the objectives of the managers. Ignition is typically with a drip torch along roads from a truck or ATV (all-terrain vehicle). Fire permits are required and are issued on a daily basis by state (Florida and Georgia) forestry departments.

Prescribed fire is used extensively to manage habitat to produce high populations of northern bobwhite in the Red Hills landscape in southern Georgia and northern Florida. A modified envirogram reveals how fire directly and indirectly influences northern bobwhite fitness (Figure 4). This envirogram only includes factors that are associated with fire. For example, the effects of fire on mate availability are unclear and therefore left blank. Resources in the centrum of the bobwhite envirogram that are influenced by fire are mainly foods: seeds, leaves and insects. Soil disturbance (harrowing) in food plots and agricultural fields and fire are used to manage the composition and structure of vegetation. Soil disturbance creates suitable substrate for weedy plant species that provide the preferred foods of quail. Woody species tend to suppress grasses (Wilson 1998). Fire frequently kills hardwood saplings or aboveground stems that would radically alter the structure and composition of the ground cover plant community, particularly the grasses. Fire frequency, severity, and season affect vegetation composition and age, which are important for herbaceous insects that are food for young bobwhite quail. Various hawks (e.g., *Accipiter* and *Buteo*) are predators of bobwhite and the bobwhite requires brush for escape cover. Patchiness of burn, although difficult to obtain predictably, could provide essential cover. The season for fire also might have a strong affect on visibility of quail to hawk predators. Eliminating cover by fire during hawk migration in the spring, when numbers of hawks are at their peak, could increase predation by hawks. On the other hand, delaying burning until later in the spring to minimize predation by migratory hawks creates the potential of increasing the loss of bobwhite nests directly to fire. The relative importance of loss of some adults to predation to loss of nests to fire to bobwhite fitness should inform management on the preferred season of fire.

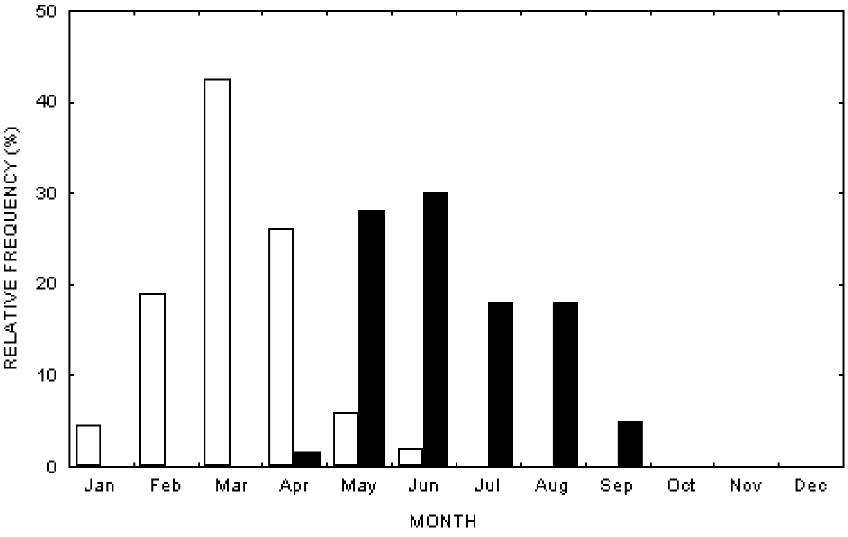

Figure 3.—Frequencies of fires started by lightning (solid bars) and prescribed fires initiated for northern bobwhite management (open bars) in the Red Hills region of north Florida and south Georgia (adapted from Brennan and others 1995 with permission).

Examination of this envirogram reveal some potential difficulties in the long-term use of fire within the framework of short-term maximization of bobwhite numbers, particularly in the interaction between soil disturbance, vegetation composition, and fire. Soil disturbance—depending on its scale and intensity—may cause the loss of native, pyrogenic grasses and forbs, particularly wiregrass in the Red Hills. Creation of food plots, small agricultural fields, and harrowing have the simultaneous effects of generating preferred foods of the bobwhite and eliminating the ground cover plant species that are best adapted to carry fire. The grasses and forbs that replace the pyrogenic species apparently are less able to compete with invasive hardwoods. The scale and severity of soil disturbance are likely to be influential in ground cover plant species recovery.

The tradeoffs between intensive management for the bobwhite and pyrogenic ground cover emerge in the direct and indirect effects of the environment on bobwhite fitness (Figure 4) and can also be seen within the framework of coarse and fine filters (Agee 1999). Management for short-term increases in bobwhite numbers (fine filter) through soil disturbance alters the ground cover composition that best supports prescribed fire. Impairing the competitive balance between grasses and woody plants through harrowing or plowing will ultimately limit the effectiveness of prescribed fire as the principal form of disturbance within the ecosystem. A coarse-filter approach in which the natural ground cover composition is left undisturbed (e.g., retention of wiregrass) and, to a lesser extent, the season of fire is shifted to the growing season within the Red Hills may be necessary to prevent ecosystem instability and potentially loss of ecosystem composition and function.

Conclusions

Use of prescribed fire has been challenged for reasons of public health and safety. Challenges have also emerged for ecological reasons, because fire has positive or negative effects on different species (e.g., long-distance migratory songbirds vs. red-cockaded woodpecker or rare butterflies vs. grassland birds). These potential sources of conflict will require increasingly sophisticated use of prescribed fire to achieve multiple ecological objectives. The tools at the disposal of the manager who employs fire (severity, season, size, etc.) need to be matched carefully with our best knowledge of the environmental influences of one or more species of interest.

- One current framework for understanding the effects of fire on birds (see Smith 2000) follows the theory of the environment developed by Andrewartha and Birch (1983). The envirogram is a useful graphical technique that captures the multiple direct and indirect effects of fire—both positive and negative—in the life history of an organism.

- The envirogram is a single-species approach that could be used in multi-species conservation planning by selecting indicator species and examining the location and nature of fire within those envirograms. The complexity of managing for multiple species suggests that a coarse-grained approach may be necessary for long-term sustainability.

- The longleaf pine ecosystem is largely a grassland that has largely been eliminated and degraded. Fire exclusion has dramatic effects on the avian communities of longleaf pine woodlands, and groundcover disturbance, habitat fragmentation, and ecotone disruption have more subtle, long-term effects on birds. Within relatively intact longleaf pine forests, the frequency, season, ignition pattern, and intensity of fire can be manipulated to obtain the desired structure and function.

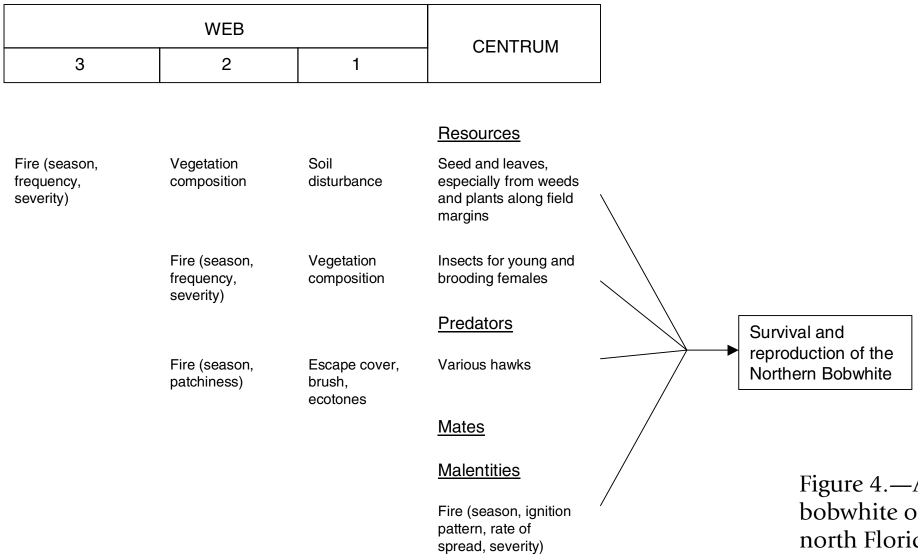

Figure 4.—An abbreviated envirogram for the northern bobwhite on hunting estates in the Red Hills region of north Florida and south Georgia that includes all occurrences of fire in the web and centrum.

Acknowledgments

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Management Options for Songbirds Using the Oak Shelterwood-Burn Technique in Upland Forests of the Southeastern United States

J. Drew Lanham[1], Patrick D. Keyser[2], Patrick H. Brose[3] and David H. Van Lear[1]

Abstract.—The shelterwood-burn technique is a novel method for regenerating oak-dominated stands on some upland sites while simultaneously minimizing undesirable hardwood intrusion with prescribed fire. Management options available within an oak-shelterwood burn regime will create variably structured habitats that may potentially harbor avian communities of mature forest and early successional species (canopy retention); grove-woodland species (post-harvest prescribed burn) or shrubland species (total harvest). We suggest that the management options associated with shelterwood-burn silviculture offer viable alternatives for managing songbird and timber resources where oak-dominated stands are the desired goal in upland southeastern sites.

Introduction

Songbirds have been the focus of many conservation efforts as declines in populations of many species, especially Neotropical migrants, have been recorded in the eastern United States (Askins et al. 1990). While numerous investigators have reported the effects of various silvicultural treatments on songbirds (e.g. Conner and Adkisson 1975; Webb et. al. 1977; Evans 1978; Crawford et. al 1981) there is a dearth of information addressing the effects (real or potential) of prescribed fire on songbirds in hardwood systems of the Southeast.

Oaks, *Quercus spp.*, are one of the most important food and cover resources for forest wildlife in the Southeast (Martin et al. 1956). A large number of songbirds, including many species of Neotropical migrants, occupy oak forest types in southeastern North America (Hamel et al. 1982). Additionally, oaks are a valuable economic commodity producing high quality timber for a variety of uses. Because of its value, oak regeneration is a priority on many upland sites. Shelterwood silviculture is widely used to regenerate oak stands on upland sites (Sander et al. 1983). It is employed so that the partial harvests will reduce the dense shade that suppresses vital root development of existing oak regeneration (Loftis 1990; Sander 1971).

By retaining the canopy and maintaining partial shade, the rapid growth of shade-tolerant species such as yellow poplar (*Liriodendron tulipifera*), is inhibited. The added benefit of litter and soil disturbance during harvest operations prepares seed beds for acorns and oak seedling establishment (Cook et al. 1998).

However, because of a paucity of oak seedlings and sprouts in mature oak stands and/or the inability of existing oak stock to out-compete other vegetation, shelterwood cuts alone are often ineffective in promoting oak regeneration (Smith 1993; Lorimer 1993). As a result expensive pretreatment measures such as herbicide application (Loftis 1990; Lorimer et al. 1994), low-intensity prescribed fires (Barnes and Van Lear 1998; McGill et al. 1999), tree shelters (Potter 1988) and nursery stock plantings (Bowersox 1993; Gordon et al. 1995; Schlarbaum et al. 1997) must be implemented 5-15 years before the initial harvest. Such treatments are unattractive to natural resource managers and private landowners with limited budgets.

A more efficient means of regenerating oak stands (hereafter referred to as the oak shelterwood-burn technique) was developed jointly in 1993 by research conducted by the Virginia Department of Game and Inland Fisheries (VDGIF) and Clemson University Department of Forest Resources (Clemson, South Carolina). By burning two oak-dominated shelterwood stands after an initial harvest (Keyser et al. 1996), the regeneration of yellow poplar, red maple (*Acer rubrum*), and sweetgum (*Liquidambar styraciflua)* was reduced by 67-90% while oak reproduction was reduced by only 11%. Subsequent studies of fire effects in oak-dominated shelterwood stands in the Virginia Piedmont (Brose and Van Lear 1998a; Brose et al. 1999) and the Northeast (Ward and Gluck 199) showed the same trends in fire resistance of oak and demonstrated the critical role of fire intensity coupled with growing-season burns for creating a cohort of tree seedlings dominated by oaks.

Although the oak-shelterwood burn technique was originally implemented to improve the viability of oak regeneration and the production of hardwood timber in uplands, the conservation of biodiversity is often a goal of many forest management initiatives. Because the shelterwood-burn method is a novel technique and could be adopted by increasing numbers of landowners and natural resource managers, it will be important to understand how wildlife communities might respond. Here, we examine how various management options implemented in oak-shelterwood burn sites could influence the composition of songbird communities. The potential influences of the management options described here are related to the vegetative structure and composition that will result from the three oak-

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[1] Department of Forest Resources, Clemson University, Clemson, SC 29634

[2] Westvaco Forest Resources, Rupert, WV 25984

[3] Northeastern Research Station, USDA Forest Service, Irvine, PA 16329

shelterwood burn options and from inferences drawn from other bird habitat studies conducted in upland oak-dominated stands where silvicultural treatments have created habitat conditions similar to those expected in oak-shelterwood burns.

The Shelterwood Burn Technique

The oak shelterwood-burn technique is a three-step process. First, an initial shelterwood cut leaves 50-60 dominant oaks per ha (11 to 12 m^2 of basal area/ha). The remnant stand of oaks is comprised of the best stock to provide a vigorous regeneration cohort. Next, the stand is left undisturbed for 3 -5 years while the regeneration layer develops. After 3-5 years, a hot (flame length > 1.0 m) growing-season fire is applied to the stand, resulting in an oak-dominated regeneration cohort. Each of these steps and the added option of complete stand harvest after the development of a strong regeneration cohort will create three variably structured forest habitats that may be used by a wide variety of resident and migrant songbirds.

Options for Songbird Management

Option 1: Canopy Retention

Canopy retention treatments provide two-age stands twice during the shelterwood burn cycle. First, during the phase when shade-tolerant hardwoods such as yellow poplar and red maple dominate the advance-regeneration pool and second, after a satisfactory cohort of vigorous, advance oak regeneration is achieved when a portion or all of the residual overstory trees may be retained for at least half of the next rotation. Retention of a partial overstory during either phase may provide sufficient canopy habitat and vertical structure for some species of mature forest birds (Dickson et al. 1995). Relative to other even-aged silvicultural methods, canopy retention treatments would be the least intensive and probably most similar to an uneven-aged mature forest.

Crawford et al. (1981) surmised that timber management strategies altered bird communities in relationship to the degree of stand disturbance. They predicted that partial harvests would provide sufficient canopy cover to buffer complete species turnover from mature forest to early-successional species observed in clearcut forests. They further stated that partial cuts would return more quickly to site conditions conducive to mature forest species than would even-aged treatments. These findings have been corroborated by a number of other studies that have shown that although populations of some forest-interior songbirds may be reduced relative to an undisturbed stand due to habitat alteration, increased nest predation, and parasitism (Webb et al. 1977; Nichols and Wood 1995), these species are generally not entirely eliminated and population recovery may occur rapidly as the new forest matures (Conner and Adkisson 1975; Askins and Philbrick 1987).

Dickson et al. (1995) support the idea that the retention of a residual canopy (<50%) for several years after an initial harvest can provide habitats for some mature forest birds that would not inhabit stands managed using traditional even-aged management technique. In West Virginia, Nichols and Wood (1995) found that two-age stands contained a greater density, richness, evenness, and overall diversity of breeding birds than early-successional and mature stands. Total density for all Neotropical migrants was also highest in the two-age stands. Densities of forest-interior species were not different between clearcut, mature, and two-age stands. The two-age stands had densities of interior-edge species equal to or greater than the other two treatments. These patterns can be explained, in part, by the occurrence in two-aged stands of species normally associated with (a) forest interiors: veery (*Catharus fuscescens)*, American redstart (*Setophaga ruticella*), and scarlet tanager (*Piranga olivacea*), and (b) early-successional habitats: chestnut-sided warbler (*Dendroica pensylvanica*), indigo bunting (*Passerina cyanea)*, and eastern towhee (*Pipilo erythrophthalmus*). The co-occurrence of mature forest and early-successional species within the same areas indicated that two-aged stands might provide both types of habitats for these species.

Annard and Thompson (1997) reported higher species richness for breeding birds in stands treated by shelterwood cuts than in clearcuts, group selection, single tree selection, or uncut stands in the Missouri Ozarks. The number of species detected were higher in shelterwoods than in uncut controls or uneven-aged stands. As with Wood and Nichol's study, these differences were attributed to the presence of a mixture of early-successional and mature forest bird species including blue-winged warbler (*Vermivora pinus*), and prairie warbler (*Dendroica discolor*). These species occupied shelterwood stands along with birds more commonly associated with mature stands such as red-eyed vireo (*Vireo olivaceous)*, worm-eating warbler (*Helmitheros vermivorous*), and Acadian flycatcher (*Empidonax virescens*).

Nesting success must be considered in conjunction with measures of density and diversity of breeding birds. Nichols and Wood (1995) did not find any differences in nest success among treatments in West Virginia. Nest parasitism by brown-headed cowbird (*Molothrus ater*) was not a major factor in their study with only eight of 246 nests parasitized and no differences in the number of cowbirds found among treatments. Annard and Thompson (1997) and Welsh and Healy (1993) found similar results in Missouri and New Hampshire, respectively. One must remain aware, however, as patterns of predation and parasitism may vary depending on the landscape context. The impact of cowbirds and predators in extensively forested systems tends to be lower than those in agricultural and suburban landscapes (Wilcove 1985).

Canopy disturbance has been shown to benefit some forest-interior bird species that have declined in some regions. Some bird species that use early-successional gaps within mature forests may decline in areas where disturbances do not produce the regenerating ground-layer and shrub vegetation they prefer (Franzreb and Rosenberg 1997). Shelterwood harvesting increases light levels and soil disturbances that stimulate the growth of low vegetative cover, i.e herbs/forbs/shrubs. In West Virginia, Nichols and Wood (1995) found that the Kentucky warbler (*Oporonis formosus*), wood thrush (*Hylocichla mustelina*), American redstart, and black-and-white warbler (*Mniotilta varia*) were 2-3 times more abundant in two-age stands than in uncut controls.

The retention of 11-12 m^2 of oak basal area/ha (50-60 dominant oaks/ha) in shelterwood stands also provides reliable acorn sources (Healy 1997). This is important because the acorns provide seed sources for regeneration. Acorns are also one of the most important wildlife food resources as they are consumed by more than 200 wildlife species throughout North America. Among these are many species of songbirds (Martin et. al. 1951; Beck 1993). Based upon the floristic structure of stands expected after canopy retention treatments, Table 1 lists bird species that are likely to occur in the diverse two-age structure of these areas.

Option 2: Shelterwood Prescribed Burning

The second option in the oak-shelterwood burn scheme is the use of periodic prescribed fire in partially harvested stands. Among the three options discussed here, this method is likely to be intermediate in its effects on the songbird community. Ultimately, the shift in species composition will vary depending on the vegetative structure that results from the season, intensity, and frequency of the prescribed burns. Dormant-season burns produce low-growing, sprouting regeneration of shrubs and trees and stimulate the production of soft mast (Stransky and Rose 1984). These responses may provide forage, cover and arthropod prey for many songbird species (Dickson 1981).

Repeat dormant-season burning increases the abundance of oak regeneration. Oak regeneration is limited by additional fires and then released at intervals by withholding burning treatments, creating patchy stands in different successional stages. Dickson (1981) surmised that in southern pine and pine-hardwood forests, a patchwork of different successional stages within a stand (or across a landscape) could enhance bird diversity and abundance. This patchwork would obviously be dependent not only upon the frequency and intensity of fires but also on the size, topography and site capability of the area burned. In stands managed with dormant season fires that will allow the proliferation of hardwood shrubs and trees underneath an open canopy , bird communities are likely to be comprised primarily of shrub nesting (e.g.white-eyed vireo, *Vireo griseus*) and midstory species (e.g.wood thrush) along with species more characteristic of open canopy forests such as the yellow-billed cuckoo,*Coccyzus americanus,* and blue-gray gnatcatcher (*Pilioptila caerulea*). More so than other burning treatments, dormant-season fires in oak-shelterwoods are likely to produce vegetative characteristics and therefore bird communities more similar to two-age canopy retention stands.

Annual or biennial prescribed burning during the growing season would create open hardwood woodlands and savannas by gradually eliminating much hardwood shrub and tree regeneration while stimulating production of ground-level herbaceous vegetation (Thor and Nichols 1973). Oak woodland and savanna habitats were described as common landscape features by early explorers and settlers who observed the Native Americans' extensive use of fire (Pyne 1982; Buckner 1983; Van Lear and Waldrop 1989). Over time, however, oak savannas and woodlands and some of the wildlife species associated with them have become rare. The restoration of hardwood savannas and open woodlands would probably shift bird guilds from mature forest-interior species to canopy and midstory dwelling, open woodland and grove species such as great-crested flycatcher (*Myiarchus crinitus*), eastern wood-pewee (*Contupus virens*), orchard oriole (*Icterus spurius*) and summer tanager (*Piranga rubra*). Although growing-season fires might benefit some bird species, others could be negatively impacted by burns initiated so late that nesting and other breeding activities are disrupted. Therefore, spring burning should be judiciously prescribed as early as possible in the season so that direct impacts on nesting or breeding birds are minimized.

Fire intensity ("hot versus cool") also affects vegetative structure and therefore avian community composition. In a study conducted in Alabama pine-hardwood Piedmont sites, Stribling and Barron (1995) found a greater abundance and diversity of birds in burned stands subjected to cool fires, with canopy, shrub and cavity nesters being most abundant. Canopy, shrub and bark feeding species were also more abundant in cool burn sites than in untreated stands. These differences were attributed to the increased heterogeneity of vegetative structure (patchiness) of treated areas. Stribling and Barron (1995) found ground-foraging and ground-nesting songbirds to be more abundant in pine-hardwood stands treated with a hot, early-spring fire than in those treated with cooler early-spring fires. They attributed this response to the removal of litter, which they hypothesized provided better foraging and nesting areas for birds in those guilds.

Some residual damage can occur from prescribed burning. Residual overstory trees, especially thin-bark

Table 1.—Neotropical migrant birds associated with oak-shelterwood burn options in upland southeastern forests.

Canopy Retention	Shelterwood Burning	Complete Harvest
yellow-billed cuckoo (*Coccyzus americanus*)		
whip-poor-will (*Caprimulgus vociferus*)	X	
chuck-will's widow (*C. carolinensis*)	X	
ruby-throated hummingbird (*Archilocus colubris*)	X	X
Acadian flycatcher (*Empidonax virescens*)		
	eastern kingbird (*Tyrannus tyrannus*)	
great-crested flycatcher (*Myiarchus crinitus*)	X	
	eastern wood-pewee (*Contupus virens*)	
gray catbird (*Dumatella carolinensis*)	X	
wood thrush (*Hylocichla mustelina*)		
veery (*Catharus fuscescens*)		
blue-gray gnatcatcher (*Pilioptila caerulea*)		
yellow-throated vireo (*Vireo flavifrons*)	X	
red-eyed vireo (*V. olivaceous*)		
		white-eyed vireo (*V. griseus*)
Blackburnian warbler (*Dendroica fusca*)		
black-throated blue warbler (*D. caerulescens*)		
black-throated green warbler (*D. virens*)		
cerulean warbler (*D. cerulea*)		
chestnut-sided warbler (*D. pensylvanica*)		X
		prairie warbler (*D. discolor*)
yellow-throated warbler (*D. dominica*)	X	
		yellow warbler (*D. petichia*)
		blue-winged warbler (*Vermivora pinus*)
		golden-winged warbler (*V. chrysoptera*)
American redstart (*Setophaga ruticilla*)		
black-and-white warbler (*Mniotilta varia*)		
		common yellowthroat (*Geothlypis trichas*)
hooded warbler (*Wilsonia citrina*)		
Kentucky warbler(*Oporonis formosus*)		
northern parula (*Parula americana*)		
ovenbird (*Seiurus aurocapillus*)		
Louisiana waterthrush (*S. motacilla*)		
worm-eating warbler (*Helmitheros vermivorus*)		
		yellow-breasted chat (*Icteria virens*)
orchard oriole (*Icterus spurius*)	X	
Baltimore oriole (*I. galbula*)	X	
scarlet tanager (*Piranga olivacea)*		
summer tanager (P. *rubra)*	X	
indigo bunting (*Passerina cyanea*)		X
		blue grosbeak (*Guiraca caerulea*)

Habitat associations inferred from "Primary Habitats" and "Key Habitat Requirements" designations by Hamel et al. (1982): Oak-shelterwood analogs are as follows: canopy retention = sapling poletimber-sawtimber; shelterwood burning =grass-forb, sawtimber; complete harvest =seedling-sapling.

species such as maples (*Acer* spp.) and yellow poplars and those with slash accumulations at their bases (Brose and Van Lear 1999) are prone to fire-kill or damage. However, the creation of dead and dying trees (snags) provide important foraging sites for woodpeckers and other bark gleaning species such as the black-and-white warbler. Snags also provide perching/hawking sites and roosting/nesting habitats. Larger sized snags are valuable nesting habitats for both primary cavity excavators (woodpeckers) and secondary cavity nesters including Neotropical migrants such as the great-crested flycatcher (Lanham and Guynn 1993). In addition to the valuable functions snags, downed logs and other coarse woody debris (e.g. tree tops, fallen limbs) provide for birds, these features provide habitat for forest-floor-dwelling arthropods, herpetofauna, and small mammals (Hanula 1996; Loeb 1996; Whiles and Grubaugh 1996) . These provide food resources for songbirds and gamebird species such as wild turkey, *Meleagris gallopavo,* and northern bobwhite, *Colinus virginianus*. Coarse woody debris also helps to prevent erosion in steep terrain by slowing overland water flow (Van Lear and Danielovich 1987) and builds soil as it decays by slowly releasing nutrients and organic matter. These actions will affect the structure and composition of vegetation and ultimately the avian community on a harvested site. Because fire in forested stands can have such varied effects, a wide variety of bird species may be supported based upon fire frequency, intensity and various site characteristics. Because most natural resource mangers and private landowners will be primarily concerned with the production of open, oak-dominated woodlands, Table 1 lists some species likely to occur in understory (growing-season) burned treatments that result in park-like oak woodlands.

Option 3: Overstory Removal

A third option is harvesting all of the residual overstory trees. This approach creates even-aged, early-successional hardwood habitat and is the most intensive of the options described here. In the initial year after harvest, stands without an overstory will undergo a dramatic turnover in species. Species such as indigo bunting and field sparrow, *Spizella pusilla,* are common in regenerating hardwood stands during these initial grass-forb and seedling-sapling stages (Evans 1978). In subsequent years tree saplings and shrubs increase the vertical structure within a regenerating stand at which point avian diversity and abundance levels can surpass those found in mature stands (Conner and Adkisson 1975; Thompson and Fritzell 1990). Thompson et al. (1993) attributed this peak in diversity and abundance to increases in vegetative vertical structure and horizontal patchiness within and among stands. In southeastern uplands, regenerating seedling-sapling hardwood habitats are preferred by shrub-scrub species such as prairie warbler, yellow-breasted chat, *Icteria virens*, and chestnut-sided warbler. Table 1 lists some Neotropical migrant bird species typical of regenerating, early-successional hardwood stands.

As regenerating stands age to form closed canopy sapling-poletimber stands, species richness and abundance frequently decreases to levels below younger shrubland and older mature forest habitats (Conner and Adkisson 1975). However, some forest-interior songbirds such as black-and-white warbler and wood thrush will begin using pole stands at this stage (Conner and Adkisson 1975; Askins and Philbrick 1987).

Discussion and Conclusions

The hardwood forests of eastern North America are one of the largest broad-leaved, deciduous ecosystems in the world (Hicks 1997). Among these ecosystems, upland oak-dominated types are among the most widespread and important as economic and ecological resources. The songbird communities dependent upon these habitats in the Southeast include a large number of Neotropical migrants (Hamel et al. 1982; Thompson and Fritzell 1990). Since many of these species are declining, the management of their habitats has become a conservation priority. Although the prevailing songbird conservation paradigm in many eastern hardwood-dominated forests has been to limit harvests to single/ group-tree selection or eliminate cutting entirely, thousands of hectares of oak-dominated forests occur on private lands where wildlife conservation goals may be secondary to timber management priorities. This means that no cut and selection cut options are often unrealistic. Innovative management is needed to satisfy both goals.

Burning as a silvicultural technique in southeastern forests has traditionally been used in pine stands. Conversely, it has been regarded as a disturbance to be prevented in hardwood forest management. The shelterwood burn technique has been shown to be an effective method for regenerating oak-dominated stands in the southeastern Piedmont (Van Lear and Brose 1999). A number of other studies conducted in two-age shelterwood systems (e.g. Nichols and Wood 1995; Annard and Thompson 1997) have proven that shelterwood techniques offer a viable option for songbird conservation/management. Additionally, Stribling and Barron (1995) showed that fire could also play a positive role in forest bird management. We suggest that stands managed in different stages of the shelterwood-burn process across a landscape would offer habitats similar to those two-age and burned stands. The result, we believe, will be a diversity of habitats attractive to forest-interior, edge-interior, open woodland and early-successional shrubland species.

While wildlife and timber production goals are frequently in opposition, the ability to reliably reproduce oak-dominated stands using a less intensive form of even-aged management like the oak-shelterwood burn technique and the associated options might prove to be a strategic tool for both wildlife conservation and sustainable timber production in southeastern uplands. We do not suggest that the oak-shelterwood burn system offers solutions for conserving every songbird species within a given stand. We do suggest, however, that this technique offers novel opportunities for sustainable timber production and effective songbird conservation across southeastern landscapes where both are management objectives.

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Influence of Ecosystem Restoration for Red-cockaded Woodpeckers on Breeding Bird and Small Mammal Communities

Ronald E. Masters[1], Christopher W. Wilson[2], Douglas S. Cram[1], George A. Bukenhofer[3] and Robert L. Lochmiller[4]

Abstract.—Shortleaf pine-bluestem (*Pinus echinata* Mill.-*Andropogon* spp.) restoration for red-cockaded woodpeckers (*Picoides borealis*) has been underway for more than 2 decades on the Ouachita National Forest, Arkansas. Restoration efforts consist of modifying stand structure to basal areas similar to presettlement times and reintroduction of fire. This is accomplished through midstory and codominant tree removal (wildlife stand improvement -WSI) and dormant season prescribed fires on a 3-year cycle. Concern has been expressed about the influence of this type of management on non-target species, specifically small mammals and breeding birds. Control stands (no thinning or fire) were characterized by closed canopies and dense midstory with little understory vegetation. WSI-treated stands were characterized by open canopies, little midstory and an increase in herbage production by 3-7 fold depending on whether or not stands had been burned and time since the stands had been burned. Woody cover after WSI followed a predictable increase with additional growing seasons since prescribed fire. Total community abundance, species richness, and diversity of small mammal and breeding bird communities were dramatically increased by restoration. No small mammal species were adversely affected by restoration treatments. Pine-grassland obligate songbirds increased following WSI and many of these showed significant increases directly attributable to fire. Bird and small mammal habitat associations are also described.

Introduction

Shortleaf pine (*Pinus echinata* Mill.) -grassland habitats were once a prevalent landscape component in the Ouachita Mountains (Nuttall 1980; Foti and Glenn 1991; Masters et al. 1995). Frequent fire maintained these woodlands as distinctly open, pine-dominated communities (Jansma and Jansma 1991; Masters et al. 1995). Fire suppression has led to the replacement of pine-grassland woodlands with closed canopy pine-hardwood forest types throughout the southeastern United States (Waldrop et al. 1992). To improve habitat conditions for the endangered red-cockaded woodpecker (see list of common and scientific names for bird species), the U. S. Forest Service began thinning midstory and some codominant pine and hardwood trees in 1979. This cultural practice, wildlife stand improvement (WSI), is followed by dormant-season prescribed fire every 3 years (U.S.D.A. 1985). In the early 1990s, application of these practices was broadened to an ecosystem approach for restoration of the shortleaf pine- bluestem (*Andropogon* spp.) community on the Ouachita National Forest.

For landscape and ecosystem restoration, quantitative knowledge of historical vegetation patterns across the landscape is essential in order to develop suitable restoration targets. Historical land use documents such as General Land Office (GLO) survey notes have successfully been used to describe presettlement and settlement landscapes (Cottam 1949, Lorimer 1980, Foti and Glenn 1991). Analysis of GLO data in the Ouachita Highlands of Arkansas (Foti and Glenn 1991) and later in Oklahoma (Kreiter 1995) provided targets for stem density, basal area and information on tree species composition for renewal of the pine-bluestem community.

Birds and small mammals have often been used as representative taxa to determine vertebrate wildlife responses to forest management practices (Morrison 1992, Masters et al. 1998). Bird distribution and abundance is largely determined by vegetation structure (MacArthur and MacArthur 1961; Wiens 1974; Roth 1976) and plant species composition (Smith 1977; Rice et al. 1983; Brennan and Morrison 1991). As well, habitat suitability for small mammals may be altered by forest management practices that modify primary production or vegetation structure (Masters et al. 1998).

The relationship between habitat attributes and bird species richness and abundance has been demonstrated in a number of forest types (e.g., Karr and Roth 1971; Willson 1974; James and Wamer 1982; Yahner 1986) and also for many small mammals (e.g., Dueser and Shugart 1978; Grant and French 1980; Root et al. 1990; Tappe et al. 1994). While several studies have examined bird and small mammal-habitat relations following fire, none we are aware of have sought to determine population response and habitat characteristics important to bird or small mammal species in restored fire-dependent pine-grassland communities other than Wilson et al. (1995) and Masters et al. (1998).

Pine-grassland restoration alters plant community structure and composition that ultimately affect the breeding bird and small mammal communities. Public concern has arisen over the impacts of management for

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[1] Department of Forestry, Oklahoma State University, Stillwater, OK

[2] The Nature Conservancy, Tulsa Oklahoma 74114

[3] USDA Forest Service, Atlanta, GA 30309

[4] Deceased

a single species on other members of the vertebrate community. Detailed information on population response and habitat relations in fire-dependent woodlands is necessary to accurately assess the impacts of WSI and prescribed fire in woodland restoration and to determine relative success of efforts. Because restoration efforts have been underway for a relatively short period of time, the plant community may be in a state of transition as additional burning cycles are accomplished and as the extent of restored stands increases. Therefore we followed up on our initial efforts (Wilson et al. 1995) to assess the breeding bird community. Our objectives were to document relative abundance in untreated and treated stands and to determine plant community structural and compositional attributes that best explained breeding bird and small mammal abundance.

Materials and Methods

Study Area

Our study stands were within a >60,000-ha area proposed for pine-bluestem restoration in the west-central Ouachita Mountains in the Poteau, Mena, and Cold Springs Ranger districts of the Ouachita National Forest, Scott County, Arkansas. Topography was mountainous with numerous east-west trending ridges and valleys. We initially focused on pine-bluestem restoration stands with active woodpecker clusters or evidence of past woodpecker occupation in the vicinity (Neal and Montague 1991), but restoration efforts have now expanded outside of previously known cluster sites. All stands consisted of second-growth forest with a history of timber management and ranged in size from 14-61 ha. The study area was previously described by Wilson et al. (1995) and Masters et al. (1996).

Experimental Design

We used a completely randomized experimental design with 5 treatments replicated ≥3 times/year over 4 years for breeding birds and 2 years for small mammals. Treatments were control (n =14); WSI-no burn (WSI; n = 14); WSI, first growing season postburn (WSIB1; n = 17); WSI, second growing season postburn (WSIB2; n = 14); and WSI, third growing season postburn (WSIB3; n = 14). In 1992, we randomly chose 12 stands from a list of treated stands (n = 61), in addition to 3 randomly chosen control stands. In 1993, all WSI and WSIB3 stands were burned in early spring and therefore, became WSIB1 stands; WSIB1 became WSIB2, and WSIB2 became WSIB3. Controls remained the same, and we added 3 WSI stands for 18 stands sampled in 1993. In 1999 and in 2000, we randomly chose 20 stands each year (4 replications per treatment). The total number of stands for breeding bird work was 73 and 30 for small mammals. Details of experimental design for small mammals are found in Masters et al. (1998).

Bird and Small Mammal Surveys

We quantified relative bird abundance in 1992 and 1993, using 6, 40-m-radius point count plots 130-150 m apart with ≥90 m between plot centers and stand boundaries within each stand (Hutto et al. 1986). Methods, abundance, and frequency data were reported in Wilson et al. (1995). In 1999 and 2000, we followed the same methodology except that we sampled 3-point count plots in each stand. Analysis of previous years data demonstrated that 3 bird plots adequately characterized birds of greatest interest and the most abundant birds in the community.

We quantified small mammals using live traps and mark-recapture techniques in winter 1992 and 1993 (27 Dec - 4 Jan). We located 80 Sherman live-traps at 15 m intervals on randomly located parallel transect lines within each stand. Methods, abundance and trap-nights of effort were reported by Masters et al. (1998).

Vegetation Composition and Structure

We measured woody species composition and forest structure in 4 circular, 5-m radius subplots within each bird point count plot from late May to early June, 1992 and 1993 to examine habitat relationships. The first subplot was located at bird point count plot center. The second subplot was placed in a random direction, 30 m from point count plot center. The remaining 2 subplots were placed 120° and 240° from the second, and 30 m from point count plot center.

We determined shrub density by counting all woody stems 0.5-1.4 m in height within each subplot. We categorized stems >1.4 m in height into 8 diameter classes. We recorded all trees, shrubs, and vines within each subplot by species. We used a 10-factor prism to estimate conifer and hardwood BA in all years (Avery 1967) at the center of each subplot. We determined coniferous and deciduous canopy heights by measuring heights of 5 random overstory trees in each category with a clinometer. We estimated percent canopy cover in all years using a spherical densiometer (Lemmon 1957).

We characterized the vertical vegetation profile in 1992 and 1993 with a 10-m pole divided into 1-m segments at 9 points within each subplot. Points lay 2.5 m apart along perpendicular lines that intersected subplot center. We extended the pole vertically through the midstory at each point. We recorded presence or absence of deciduous, coniferous, and dead vegetation touching the pole for each 1-m segment.

We measured herbaceous and woody biomass by the harvest method (Cook and Stubbendieck 1986) over a 2-week period from late July to early August using 30 1-m^2 plots at 30-m intervals along randomized transects in each stand (after Masters 1991). To avoid bias caused by

Table 1.—Stand characteristics after Wildlife Stand Improvement (WSI) and growing seasons since burned on Ouachita National Forest, Arkansas, May 1992 and 1993[1]

	Treatment[2]					
	CONTROL	WSI	WSIB1	WSIB2	WSIB3	
Parameter	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	P>F
Canopy cover (%)	84.1(1.3) a	65.1(2.8) b	67.9(1.9) b	70.4(1.9) b	69.3(2.4) b	<0.001
Conifer BA (m^2/ha)	19.5(0.4) a	13.5(1.7) c	14.8(1.1) bc	17.3(0.7) ab	16.8(0.9) abc	0.015
Hardwood BA (m^2/ha)	6.9(0.3) a	4.4(0.9) ab	3.9(0.9) b	2.6(0.9) b	2.9(1.1) b	0.042
Total BA (m^2/ha)	26.4(0.4) a	17.9(1.1) b	18.7(0.9) b	20.0(0.8) b	19.7(1.3) b	<0.001
Pine canopy ht. (m)	19.5(1.0)	18.2(0.9)	20.2(0.7)	20.3(0.9)	19.3(0.9)	0.559
Hdwd canopy ht. (m)	13.7(0.7)	14.8(1.0)	14.7(1.0)	11.5(2.0)	12.2(1.0)	0.308
Tree diversity[3]	1.4(0.1)	1.1(0.2)	0.8(0.1)	0.7(0.1)	0.6(0.2)	0.072
Tree spp. Richness	9.2(1.4)	6.2(1.3)	5.6(1.0)	5.7(0.8)	5.0(0.9)	0.332
Shrub diversity[3]	2.6(0.1) b	2.6(0.1) b	2.4(0.1) b	2.8(0.1) a	2.9(0.1) a	0.001
Shrub spp. Richness	33.2(2.9) ab	36.5(3.7) a	24.9(2.7) b	34.5(1.5) ab	37.7(1.8) a	0.020

[1] Row means followed by the same letter or without letters were not significantly different at the 0.05 level (LSD).
[2] Control = no treatment; WSI = wildlife stand improvement, no burn; WSIB1 = wildlife stand improvement, first growing season following burn; WSIB2 = wildlife stand improvement, second growing season following burn; WSIB3 = wildlife stand improvement, third growing season following burn.
[3] Shannon-Weaver Diversity Index.

influence from adjacent stands, we did not sample within 50 m of stand boundaries. We clipped current year's growth of all vegetation to <2.5 cm height and hand separated into grass, sedge, panicum, non-legume forb, legume, and woody categories. We dried samples to constant weight at 70° C in a forced-air oven before weighing. Standing crop data were reported by Masters et al. (1996). We grouped data into grasslike, forb, and woody categories for bird and small mammal habitat analysis.

Data Analyses

We used the Kruskal-Wallis nonparametric test to detect treatment differences ($P < 0.05$) in stem densities by diameter class, basal areas, canopy heights, percent canopy cover, understory standing crop, and diversity measures. We analyzed data across years with stand (year x treatment) type III mean square as the error term, and used the least significant difference (LSD) test to separate mean ranks ($P < 0.05$)(Conover and Iman 1981). To further differentiate treatment effects on breeding bird and small mammal indices of relative abundance, we used specific orthogonal contrasts to test for differences between control, WSI, and burned and unburned stands. To better determine effects of pine-grassland restoration on breeding bird community, we grouped species into 3 designations of habitat occurrence: forest interior, forest edge (including some open forest species), and pine-grassland (see Wilson et al. 1995). For community level analysis we calculated Morisita's Index of Similarity (after Krebs 1989) to determine comparative community similarity between treatments. Because unequal sampling effort can bias Morisita's Similarity indices particularly with small sample sizes (Krebs 1989), we randomly selected 3 point count plots to discard for 1992 and 1993 to maintain equal sampling effort among all years in sampled stands.

We performed canonical correspondence analysis (CCA; ter Braak 1986, 1987) with the program CANOCO (ter Braak 1988) to determine the influence of 16 physiognomy and 21 floristic measures derived from vegetation samples on abundances of 27 bird species (with ≥ 1 individual/40 ha, Wilson 1994). We used Pearson product-moment correlations to examine the relationships of these same physiognomy and floristic variables with small mammal population and community parameters because CCA results were inconclusive (Masters et al. 1998).

Results

Stand Characteristics

Midstory removal (WSI) reduced percent canopy cover, conifer BA, and total BA (Table 1). Hardwood BA was lower in burned stands than in controls. Pine canopy height and hardwood canopy height were similar among controls and treatments. Shrub diversity (stems <8 cm dbh) was greatest in WSIB2 and WSIB3 stands. Tree diversity (stems >8 cm dbh) was similar among treatments (Table 1). Stand characteristics in 1999 and 2000 were not different than for 1992 and 1993.

Table 2.—Average woody stem density (stems/ha) after WSI and fire on Ouachita National Forest, Arkansas, summer 1992 and 1993[1]

	Treatment					
	CONTROL	WSI	WSIB1	WSIB2	WSIB3	
Height class, diameter class	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	$P > F$
0.5-1.4 m	2,155(432)b	10,638(2,996)a	4,647(1,279)b	11,157(826)a	2,070(1,966)a	<0.001
>1.4 m						
<1 cm	1,002(255) ab	784(258) b	94(27) c	478(81) b	1,550(279) a	<0.001
1.1-2.5 cm	218(40) a	46(11) b	14(8) c	16(6) c	112(50) ab	<0.001
2.6-8.0 cm	703(57) a	71(17) b	26(5) c	63(30) bc	92(34) bc	<0.001
8.1-15 cm	351(24) a	30(12) b	20(5) b	31(8) b	41(23 b	<0.001
15.1-23 cm	114(9) a	42(9) c	70(16) bc	96(19) ab	61(11) bc	0.005
23.1-30 cm	115(20)	71(6)	86(16)	95(16)	87(21)	0.548
30.1-38 cm	107(5)	88(17)	93(14)	99(12)	106(19)	0.881
>38 cm	39(4)	43(7)	31(7)	33(7)	34(6)	0.763
Total >1.4 m	2,648(231) a	1,175(277) b	434(22) c	910(69) b	2,082(303) a	<0.001

[1] Row means followed by the same letter or without letters were not significantly different at the 0.05 level (LSD).

[2] Control = no treatment; WSI = midstory removal within the past 2 years and no burning of downed trees; WSIB1 = midstory removal, late winter prescribed burn, in first growing season; WSIB2 = midstory removal, late winter prescribed burn, in second growing season; WSIB3 = midstory removal, late winter prescribed burn, in third growing season.

Density of woody stems <1.4 m increased in WSI stands as a result of sprouting (Table 2). Stem densities in all categories from 1.1-23.0 cm dbh decreased ($P < 0.05$) following WSI. The application of fire following WSI reduced stem densities for all diameter classes <8 cm dbh ($P < 0.05$). However, stem densities 0.5-1.4 m in height increased in WSIB2 and WSIB3 compared to controls (Table 2). Fire did not affect densities of stems >8 cm dbh.

In the vertical profile, WSI reduced total vegetation in all levels 1-10 m (Figure 1). Wildlife stand improvement reduced occurrence of conifers up to 6 m and reduced hardwood occurrence in the 1-9 m strata ($P < 0.05$). Dead vegetation increased from 0-1 m in all treated stands. Fire reduced occurrence of hardwoods and slash from 0-1 m and conifers from 1-3 m. By the third season postburn, hardwood vegetation from 0-3 m increased, largely from resprouting oaks and hickories (Figure 1).

Wildlife stand improvement increased standing crop of grasslikes, forbs, and woody vegetation <1.4 m by 3-7 fold depending on whether the stand had been burned and the time since burned (Masters et al. 1996). Woody shrubs initially decreased following fire, but became the dominant component of the understory by the third season postburn (Masters et al. 1996).

Breeding Bird Response

We observed a total of 68 species of breeding birds in all stands over the course of our study. Community composition was different between WSI treated stands and control stands in all years except 1992 (Figure 2 and 3). For all years combined, burned stands were > 97 percent similar in bird species composition. Comparison of the bird community in controls versus that in individual treatments were < 84 percent similar for all years combined. WSIB2 and WSIB3 stands had the most similar bird communities consistently in individual years and in all years combined (Figure 2).

Forest Interior

The black and white warbler, ovenbird and scarlet tanager declined in density ($P < 0.05$) in treated stands from thinning. Only the former 2 species had declines directly attributable to fire ($P < 0.05$) (Table 3). Although a tendency for decline in density was observed for the whip-poor-will, its frequency of occurrence within treated stands was significantly ($P < 0.05$) less than in untreated controls.

Forest Edge

American crow was the only species in this group to decline ($P = 0.033$) from WSI (Table 3). We found

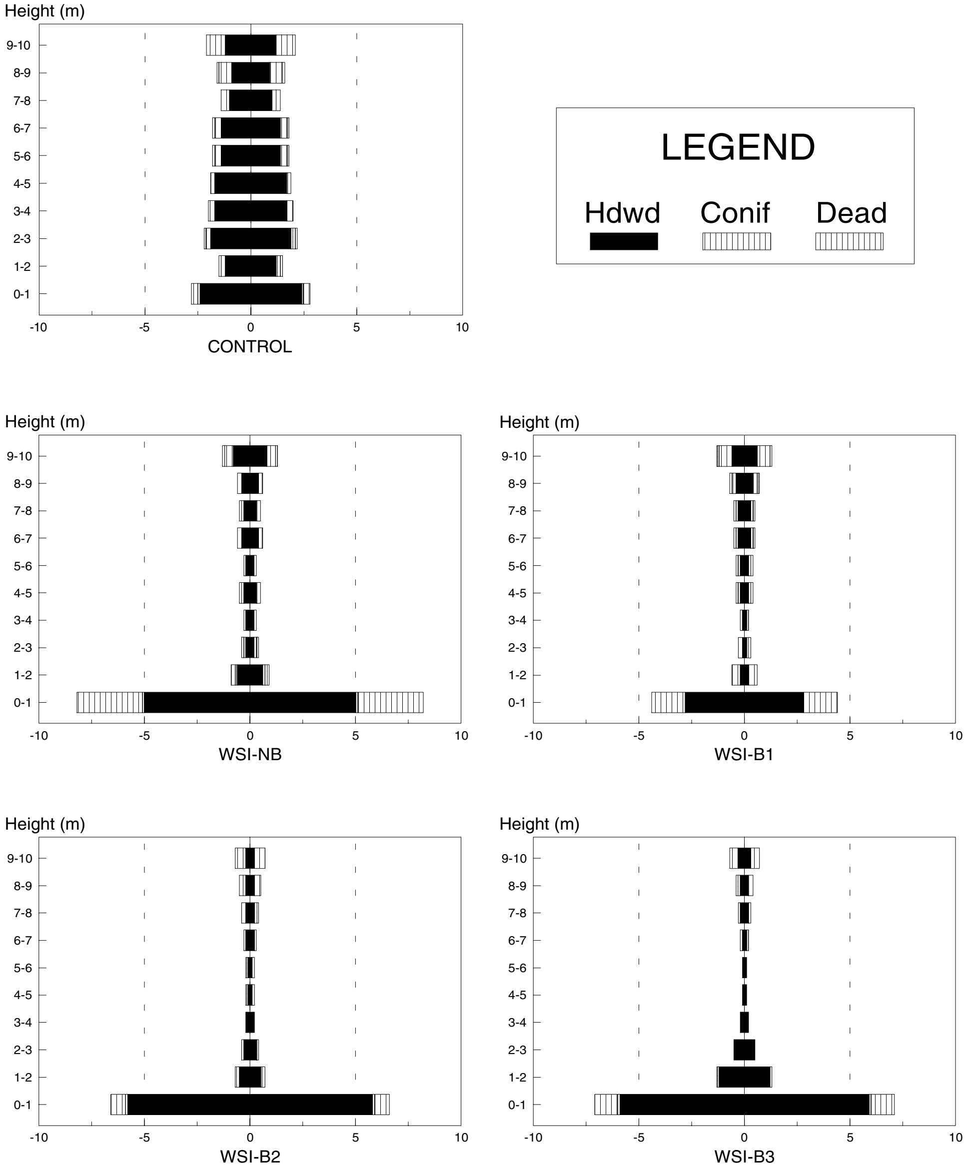

Figure 1.—Vertical vegetation profile measured within bird census plots of wildlife stand improvement (WSI)-treated and untreated stands on the Ouachita National Forest, Arkansas, May 1992-1993. Control = no treatment; WSI = midstory removal (thinning) and without burning of logging slash; WSIB1 = midstory removal, late-winter prescribed burn, following the first growing season after fire; WSIB2 = midstory removal, late-winter prescribed burn, following the second growing season after fire; WSIB3 = midstory removal, late-winter prescribed burn, following the third growing season after fire;

American goldfinch, blue-gray gnatcatcher, brown-headed cowbird, Carolina wren, white-breasted nuthatch and yellow-breasted chat density increased ($P < 0.05$) in response to WSI (Table 3). Many of these species including the common yellowthroat ($P = 0.037$) responded favorably to the addition of fire in WSI treated stands with the exception of the Carolina wren. This species increased with thinning ($P = 0.014$) then declined following the addition of fire ($P = 0.035$).

Pine-grassland

All pine-grassland species increased ($P < 0.05$) in density following either WSI or, WSI and fire except Bachman's sparrow and northern bobwhite (Table 3). However, frequency of occurrence within treated stands for these 2 species was significantly higher ($P < 0.05$) than in control stands.

Small Mammal Response

We observed a total of 9 small mammal species in all stands over the course of our study. Community composition was different between WSI treated stands and control stands in all years (Masters et al 1998). Total community richness, diversity and abundance were lowest in untreated stands. White footed-mouse (see list for scientific names) was the most dominant species and short-tailed shrew and harvest mouse were the next most abundant species (Masters et al. 1998). The white-footed mouse and short-tailed shrew occurred in all stands (Figure 4, Masters et al. 1998). Also occurring but less frequently were golden mouse, eastern woodrat, cotton mouse, hispid cotton rat, pine vole, and deer mouse listed in order of total captures (Masters et al. 1998). The WSI and WSIB1 treatments had the greatest number of species (Figure 4). We found that no species declined in response to either thinning or fire (Masters et al. 1998).

Bird-habitat Relationships

When CCA was constrained to 16 explanatory physiognomy variables, eigenvalues for axes 1 and 2 explained 48.7% of cumulative variance of bird-physiognomy relations ($P < 0.01$) (Wilson 1994). We identified 7 variables ($P < 0.05$) that explained 46% of variation in breeding bird abundances. They were: relative presence of midstory foliage (2-10 m) (16%), relative presence of foliage 0-2 m (8%), density of hardwoods > 23.1 cm dbh (5%), percent canopy cover (5%), density of all woody stems >1.4 m in height (4%), density of all woody stems < 1.4 m in height (4%), and density of conifer trees > 23.1 cm dbh (4%).

When ordination was constrained to 21 floristic variables, eigenvalues for axes 1 and 2 explained 40.1% of cumulative variance in bird species data (Wilson 1994). We identified 4 variables that explained ($P < 0.05$) 35% of variation in breeding bird abundances.

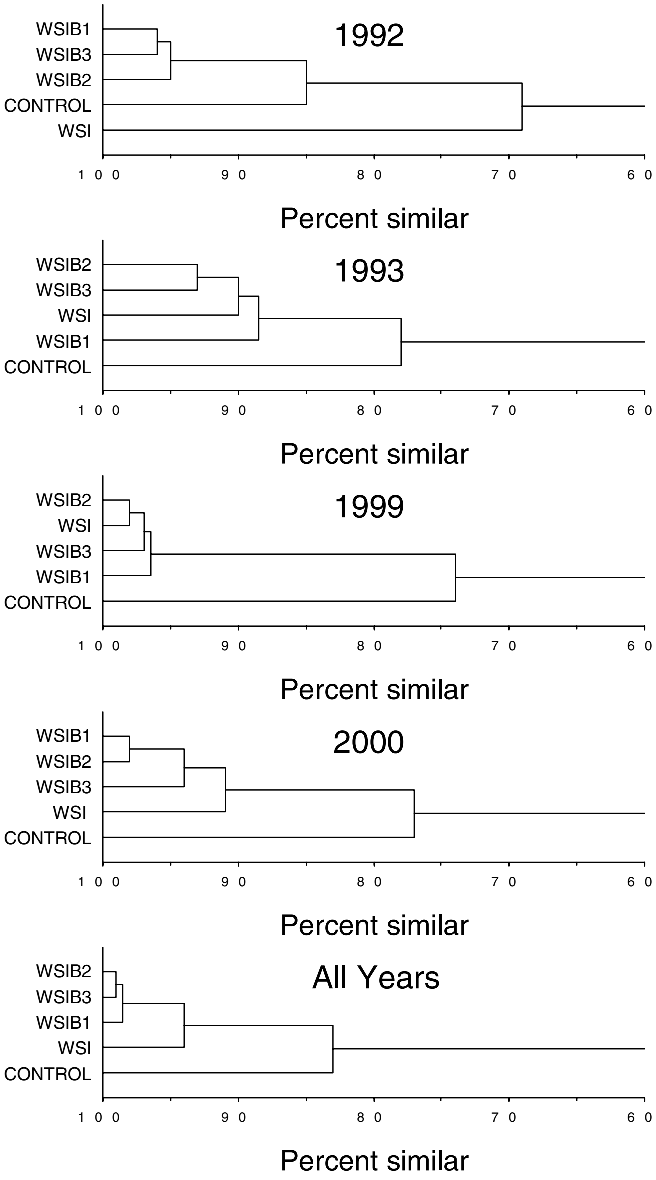

Figure 2.—Bird community similarity (Morisita's Index of Similarity) among treated versus untreated stands on the Ouachita National Forest, Arkansas, May 1992-1993 and 1999-2000. Control = no treatment; WSI = midstory removal (thinning) and without burning of logging slash; WSIB1 = midstory removal, late-winter prescribed burn, following the first growing season after fire; WSIB2 = midstory removal, late-winter prescribed burn, following the second growing season after fire; WSIB3 = midstory removal, late-winter prescribed burn, following the third growing season after fire.

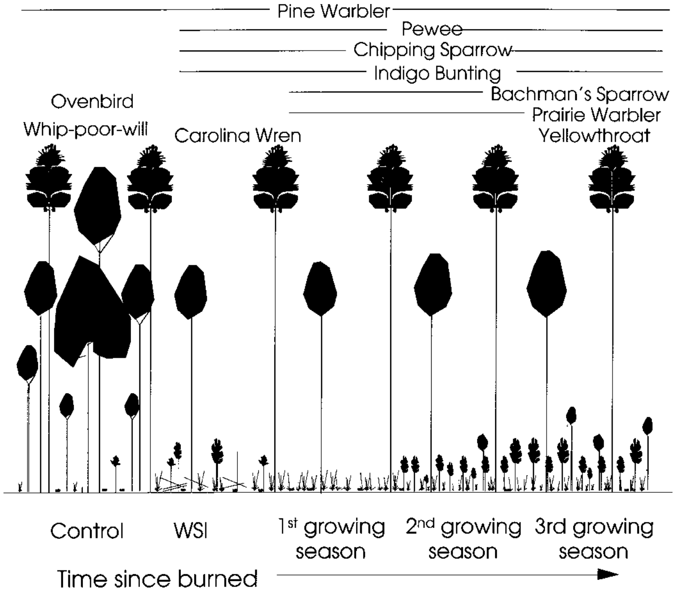

Figure 3.—Bird community model of the response of important bird species illustrating succession windows of suitable habitat among restored versus untreated stands on the Ouachita National Forest, Arkansas, May 1992-1993 and 1999-2000. Control = no treatment; WSI = midstory removal (thinning) and without burning of logging slash; 1st growing season = midstory removal, late-winter prescribed burn, following the first growing season after fire (WSIB1); 2nd growing season= midstory removal, late-winter prescribed burn, following the second growing season after fire(WSIB2); 3rd growing season= midstory removal, late-winter prescribed burn, following the third growing season after fire (WSIB3).

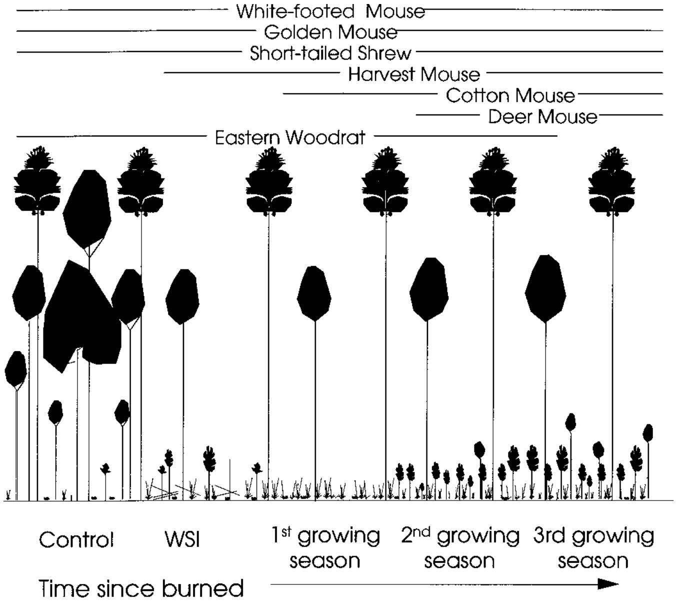

Figure 4.—Small mammal community model of the response of most species and general habitat relationships, illustrating succession windows of suitable habitat among restored versus untreated stands on the Ouachita National Forest, Arkansas, May 1992-1993. Control = no treatment; WSI = midstory removal (thinning) and without burning of logging slash; 1st growing season = midstory removal, late-winter prescribed burn, following the first growing season after fire (WSIB1); 2nd growing season= midstory removal, late-winter prescribed burn, following the second growing season after fire(WSIB2); 3rd growing season= midstory removal, late-winter prescribed burn, following the third growing season after fire (WSIB3).

They were: Shannon diversity index of woody stems > 8 cm dbh (16%), number of oak (*Quercus* spp.) stems > 8 cm dbh (9%), current annual growth of woody biomass < 1.4 m in height (5%), and the number of pine trees > 8 cm dbh (5%).

In the physiognomy CCA, variables most correlated with axis 1 were percent canopy cover, density of hardwoods > 23.1 cm dbh, midstory foliage, and density of woody stems >1.4 m in height (Wilson 1994). This axis separated species dependent on closed canopy and dense midstory forest from species requiring open canopy and fewer trees. Variables most correlated with axis 2 were number of pine trees > 8 cm dbh, density of all woody stems < 1.4 m in height, and relative presence of foliage 0-2 m. In addition to species demonstrating affinities for numbers of large conifers, this axis separated species relating positively to understory hardwood structure from species preferring little or no understory hardwood structure (Wilson 1994).

In the floristics CCA, large oaks and tree diversity were most correlated with axis 1. This axis separated species with affinities for density and diversity of hardwood trees from species preferring fewer oaks and lower tree diversity. Variables most correlated with axis 2 were current annual growth of woody biomass < 1.4 m in height and the number of pine trees > 8 cm dbh. In addition to identifying species with affinities for numerous large pines, this axis separated species according to preference for woody understory biomass. We were able to distinguish pine-grassland bird species on CCA biplots by creating a pine-grassland variable by combining large pine tree density and grass biomass (Wilson 1994).

Forest interior species

Forest interior species tended to relate positively to percent canopy cover, midstory foliage, overstory trees, and tree species diversity. The black-and-white warbler and ovenbird were strongly related to midstory foliage, percent canopy cover, and oaks >8 cm dbh in the biplots (Wilson 1994).. Scarlet tanager occurred infrequently, but was clearly related to hardwoods. Summer tanager related strongly to tree diversity in the floristic biplot (Wilson 1994).

Table 3.—Breeding bird responses (birds/40 ha)to wildlife stand improvement (WSI) and prescribed fire in pine-oak stands on the Ouachita National Forest, Arkansas, May 1992, 1993, 1999 and 2000[1]

	Treatment[2]					$P > F$		
	CONTROL	WSI	WSIB1	WSIB2	WSIB3		Contrasts[3]	
Group, Species	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Trt	C vs WSI	NB vs B
Forest Interior								
Acadian Flycatcher	0.9(0.5)	0.6(0.4)	0.3(0.3)	1.3(0.9)	1.3(0.9)	0.485	0.232	0.666
American Redstart	0.0(0.0)	0.6(0.6)	0.5(0.5)	0.0(0.0)	0.0(0.0)	0.481	0.456	0.339
Black & White Warbler	13.3(3.6) a	9.2(2.4) ab	1.6(1.3) c	3.2(1.3) c	4.4(1.8) bc	0.009	0.010	0.012
Broad-winged Hawk	0.3(0.3)	1.6(1.3)	0.0(0.0)	0.3(0.3)	0.0(0.0)	0.444	0.734	0.129
Carolina Chickadee	8.8(3.4)	8.2(5.3)	10.7(4.1)	8.2(3.4)	3.8(1.7)	0.868	0.445	0.799
Cooper's Hawk	0.0(0.0)	0.6(0.6)	0.0(0.0)	0.0(0.0)	0.3(0.3)	0.596	0.511	0.528
Downy Woodpecker	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.6(0.4)	0.9(0.9)	0.535	0.508	0.394
Great-crested Flycatcher	3.5(1.5)	8.5(3.4)	6.2(2.1)	10.7(3.0)	2.8(1.1)	0.135	0.215	0.764
Hairy Woodpecker	0.6(0.4)	1.9(1.0)	0.5(0.4)	0.0(0.0)	0.9(0.7)	0.421	0.692	0.166
Kentucky Warbler[4]	0.3(0.3)	0.6(0.6)	0.8(0.8)	0.6(0.6)	0.0(0.0)	0.903	0.588	0.705
Ovenbird	12.6(3.7) a	0.0(0.0) b	0.0(0.0) b	0.0(0.0) b	0.0(0.0) b	<0.001	<0.001	1.000
Pileated Woodpecker	3.8(1.7)	2.2(1.1)	1.0(0.6)	0.9(0.7)	2.5(1.4)	0.656	0.264	0.347
Red-bellied Woodpecker	0.0(0.0)	0.0(0.0)	0.3(0.3)	0.3(0.3)	0.3(0.3)	0.397	0.225	0.124
Red-eyed Vireo	13.9(3.6)	16.7(3.7)	10.9(2.3)	16.7(4.0)	10.1(3.1)	0.512	0.931	0.258
Ruby-throated Hummingbird	0.3(0.3)	0.9(0.7)	1.3(0.7)	0.3(0.3)	1.3(0.9)	0.934	0.604	0.798
Scarlet Tanager	2.8(1.1)	1.6(1.6)	0.8(0.6)	1.9(1.9)	1.3(0.9)	0.193	0.023	0.895
Summer Tanager	8.5(2.3)	12.6(2.6)	9.6(2.9)	12.0(2.7)	10.1(2.7)	0.606	0.627	0.358
Tufted Titmouse	8.2(3.0)	4.1(1.6)	7.3(2.7)	0.9(0.9)	3.5(1.8)	0.063	0.076	0.652
Whip-poor-will[4]	0.9(0.7)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.414	0.061	1.000
Forest Edge and Other								
American Crow	3.8(1.5)	0.3(0.3)	1.0(0.6)	1.3(1.3)	0.9(0.7)	0.221	0.033	0.582
American Goldfinch	2.2(1.4) c	4.7(1.9) bc	14.6(4.0) ab	12.0(3.5) ab	19.3(4.7) a	0.030	0.013	0.041
Blue Grosbeak	0.0(0.0)	0.0(0.0)	0.5(0.5)	0.0(0.0)	0.0(0.0)	0.460	0.651	0.559
Blue-gray Gnatcatcher	0.3(0.3) b	5.4(1.7) a	4.7(1.6) a	0.6(0.4) b	1.9(1.4) b	0.002	0.009	0.006
Blue Jay	5.4(2.3)	3.5(1.9)	1.3(1.1)	0.6(0.4)	1.9(1.3)	0.387	0.206	0.163
Brown-headed Cowbird	0.9(0.9) b	6.6(1.8) a	8.3(2.7) a	7.6(2.3) a	2.8(1.2) ab	0.041	0.009	0.439
Brown Thrasher	0.6(0.6)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.492	0.082	1.000
Catbird	0.0(0.0)	0.3(0.3)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.402	0.610	0.065
Carolina Wren	5.1(2.1) b	49.6(8.2) a	14.0(5.0) b	12.6(4.4) b	14.8(4.6) b	0.001	0.014	<0.001
Cedar Waxwing	0.0(0.0)	0.0(0.0)	0.3(0.3)	0.0(0.0)	0.0(0.0)	0.364	0.616	0.517
Common Flicker	0.0(0.0)	0.0(0.0)	1.3(0.7)	1.3(0.7)	0.6(0.6)	0.290	0.231	0.128
Common Grackle	0.0(0.0)	0.0(0.0)	0.3(0.3)	0.0(0.0)	0.0(0.0)	0.523	0.672	0.585
Common Yellowthroat	0.0(0.0)	0.9(0.9)	0.8(0.6)	3.2(1.5)	2.8(1.7)	0.119	0.060	0.204
Mourning Dove	0.0(0.0)	0.0(0.0)	1.6(1.1)	0.0(0.0)	0.6(0.6)	0.363	0.409	0.290
Northern Cardinal	1.3(0.9)	1.9(1.4)	0.8(0.6)	0.0(0.0)	1.3(1.3)	0.702	0.575	0.280
Red-shouldered Hawk	0.0(0.0)	0.0(0.0)	0.5(0.5)	0.0(0.0)	0.0(0.0)	0.460	0.651	0.559
Turkey Vulture	0.0(0.0)	0.6(0.4)	0.3(0.3)	0.0(0.0)	0.0(0.0)	0.497	0.552	0.107
White-breasted Nuthatch	0.3(0.3)	5.7(2.1)	5.2(2.0)	7.3(1.9)	3.5(1.7)	0.146	0.032	0.881
White-eyed Vireo	0.0(0.0)	0.3(0.3)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.402	0.610	0.065
Wild Turkey	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.6(0.6)	0.402	0.610	0.511
Yellow-billed Cuckoo	0.6(0.4)	2.8(1.9)	1.3(0.6)	2.8(1.4)	1.9(1.1)	0.811	0.324	0.653
Yellow-breasted Chat	0.0(0.0) c	2.2(1.2) bc	0.5(0.5) bc	4.7(2.2) ab	8.2(2.7) a	0.007	0.019	0.360
Yellow-throated Vireo	0.3(0.3)	0.0(0.0)	0.8(0.6)	0.9(0.7)	1.3(0.9)	0.852	0.833	0.293
Pine-grassland								
Bachman's Sparrow[4]	0.0(0.0)	0.6(0.6)	0.3(0.3)	1.9(1.0)	2.5(2.0)	0.217	0.125	0.525
Brown-headed Nuthatch[4]	0.0(0.0) b	0.0(0.0) b	10.1(4.0) a	6.3(2.7) a	4.4(2.6) ab	0.042	0.068	0.023

Continued

Table 3.—continued

	Treatment[2]					$P > F$		
	CONTROL	WSI	WSIB1	WSIB2	WSIB3		Contrasts[3]	
Group, Species	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Trt	C vs WSI	NB vs B
Chipping Sparrow	1.3(1.0) b	28.1(8.5) a	43.9(6.5) a	38.2(5.8) a	35.4(7.8) a	0.003	<0.001	0.071
Eastern Wood Pewee	0.0(0.0) c	11.1(3.3) ab	12.7(2.1) a	10.1(1.8) ab	6.0(2.2) b	<0.001	<0.001	0.952
Indigo Bunting	4.7(2.3) c	57.2(9.2) a	36.4(4.5) b	54.0(6.7) a	64.1(7.3) a	<0.001	<0.001	0.699
Northern Bobwhite	0.0(0.0)	2.8(1.3)	0.0(0.0)	0.6(0.6)	2.2(1.4)	0.098	0.156	0.064
Pine Warbler	61.3(8.1)	83.7(9.8)	93.4(7.3)	102.0(7.5)	96.6(7.3)	0.115	0.017	0.272
Prairie Warbler[4]	0.0(0.0) b	9.2(2.6) a	1.6(1.1) b	7.6(2.5) a	14.2(5.1) a	0.005	0.004	0.414
Red-cockaded Woodpecker[4]	0.0(0.0) b	0.0(0.0) b	1.8(1.0) a	2.2(1.4) a	0.9(0.7) a	0.048	0.053	0.017
Red-headed Woodpecker[4]	0.0(0.0)	3.2(2.2)	2.6(1.1)	2.5(1.4)	1.6(0.9)	0.183	0.022	0.776

[1] Column means within rows followed by the same letter or without letters were not significantly different at the 0.05 level (LSD).

[2] Control=no treatment; WSI= midstory removal and no burning of downed logging slash or trees; WSIB1=midstory removal, late-winter prescribed burn, following first growing season after fire; WSIB2=midstory removal, late-winter prescribed burn, following first growing season after fire; WSIB3=midstory removal, late-winter prescribed burn, following first growing season after fire.

[3] Specific orthogonal contrasts: C=control, WSI=midstory removal, NB=nonburned WSI stands, B=WSI stands burned within previous 3 years.

[4] Priority species (Pashley et al. 2000).

Carolina chickadee was positively related to tree diversity, percent canopy cover, and midstory foliage in the biplots (Wilson 1994). The pileated woodpecker was positively related to percent canopy cover and large trees in the CCA biplots (Wilson 1994). The red-eyed vireo, tufted titmouse, and acadian flycatcher were positively related to large hardwoods and tree diversity (Wilson 1994).

Forest edge species

Species associated with forest edge or canopy openings were tolerant of wide ranges of canopy closure. These species frequently occurred at intermediate levels of midstory foliage and tree diversity. Blue jay was positively related to percent canopy cover in the physiognomy biplot. White-breasted nuthatch, yellow-billed cuckoo, and great-crested flycatcher occurred near the center of both CCA biplots, indicating preferences for intermediate values of physiognomy and floristic measures (Wilson 1994). Carolina wren was positively related to foliage 0-2 m and woody biomass <1.4 m, but negatively related to pines >8 cm dbh. Wrens benefited from the resulting slash from WSI (Wilson 1994).

The brown-headed cowbird was negatively related to percent canopy cover, oaks >8 cm dbh, and midstory foliage. Biplot positions for common yellowthroat and yellow-breasted chat were greater than all other species with respect to stems <1.4 m, foliage 0-2 m, and woody biomass <1.4 m (Wilson 1994). American goldfinch also favored open canopy structure (Wilson 1994).

Pine-grassland species

Pine-grassland species benefited most directly from reduced canopy cover, reduced midstory, and associated dense understory (Wilson 1994). We observed several species associated with sites characterized by dense understory and few trees. Prairie warbler and indigo bunting demonstrated affinities for woody understory vegetation with an aversion to overstory trees. These species were negatively related to midstory foliage and percent canopy cover. We found that pine warbler was negatively related to tree diversity and midstory foliage, and positively related to pines >8 cm dbh. Eastern wood pewee appeared to key on scattered mature trees with an open midstory. Pewees were negatively related to stems >1.4 m, midstory foliage, tree diversity, and oaks >8 cm dbh in CCA biplots (Wilson 1994).

The brown-headed nuthatch and chipping sparrow demonstrated negative relationships to stems >1.4 m, hardwoods >23.1 cm dbh, and tree diversity (Wilson 1994). Chipping sparrow was negatively associated with stems <1.4 m and foliage 0-2 m. We found red-headed woodpecker was negatively related to percent canopy cover and midstory foliage and Bachman's sparrow was negatively related to percent canopy cover, midstory foliage, oaks >8 cm dbh, and tree diversity (Wilson 1994).

Small Mammal Habitat Relationships

Most small mammal species were negatively correlated with the presence of overstory and positively associated with total biomass of understory production (Masters et al. 1998). Total community abundance and white-footed mouse abundance was positively associated with dead woody debris and woody biomass in the understory. However, total community abundance was negatively associated with the presence of midstory and more dense overstory (Masters et al. 1998). Both harvest mice and golden mouse were positively associated with the herbaceous component in the understory. As herbaceous biomass increased so did both of these species in addition to species richness and diversity (Masters et al. 1998).

Discussion

Stand Characteristics

Forests of the Ouachita Mountains at settlement (ca. 1840 in Arkansas; ca. 1890 in Oklahoma) were characterized by low-density open stands with scattered large trees (Foti and Glenn 1991; Kreiter 1995). These conditions were the result of the climatic (Foti and Glenn 1991) and Native American fire regime (Masters et al. 1995). Stands currently undergoing restoration on the Ouachita National Forest may have nearly twice the stem density and one third more basal area as open pine woodlands at settlement.

Wildlife stand improvement is an effective and timely means of recreating open woodland structure because periodic prescribed fire alone may take decades longer to alter midstory structure in a similar fashion. Following WSI, stands are characterized by pine-dominated overstory, open midstory, and dense understory of slash, hardwood sprouts, vines, grasses, and forbs. The subsequent application of fire at 3-year intervals appears to maintain this open midstory structure, at least in the short-term, because fire has its greatest impact on residual stems <8 cm dbh. Whereas understory stem densities increase with time postburn, fire top-kills small-diameter sprouts, maintaining an open midstory. Understory structure and composition will vary according to soils, slope, aspect, management history, fire frequency and intensity, and time since burned, thereby making available a range of habitats.

Bird Community

Bird community composition changed with alteration of forest structure following WSI and fire (Figures 2 and 3). Analysis of bird species relationships with physiognomy and floristic variables reveals species habitat preferences across the broad continuum of closed canopy pine-oak forest to open pine-grassland woodlands. Several species were distinctly associated with the woody structure that developed as plant community succession progressed following fire. In effect, a brief succession window of suitable habitat was created for some species such as the common yellowthroat and slightly longer windows for species with broader habitat requirements (e.g., eastern wood pewee, chipping sparrow, and indigo bunting) (Figure 3).

Seven species that occurred in our stands are listed as priority species (Table 3; Pashley et al. 2000). However, whip-poor-will was the only priority species that showed even a tendency to be detrimentally affected by restoration treatments in terms of frequency of occurrence within the stands. The Kentucky warbler was not affected. The remaining five priority species were pine-grassland obligates and were distinctly benefited by restoration treatments either in increased density or frequency of occurrence (Tables 3 and 4).

Bird-habitat relationships

Forest interior species

Mature closed canopy forest habitats are necessary for many interior forest species. Removal of midstory structure followed by prescribed fire reduces densities and diversity of midstory hardwoods, adversely affecting species dependent on closed canopies or hardwood midstory. Ground nesting species such as ovenbird and black-and-white warbler may be adversely affected by changes in understory particularly by the loss of the litter layer following fire. Other forest interior species may persist along riparian zones that dissect pine-bluestem sites, but most will decline in thinned and burned stands, particularly where hardwoods have virtually been eliminated (Wilson et al. 1995). Thinning and fire will not eliminate hardwood dependent species such as red-eyed vireo if scattered mature hardwoods are retained (Wilson et al. 1995).

Forest edge species

Tolerance of changes in forest structure varies among forest edge species. Forest edge species of mixed forest types (i.e., blue jay), will persist in stands retaining hardwoods. Shrub-dwelling species such as Carolina wren and yellow-breasted chat should occur in open situations not recently burned. Whereas periodic fire should benefit these species by promoting a shrubby understory, annual burns may adversely affect habitat. Ecotones created between restored pine-grassland woodlands and closed canopy forest may provide suitable habitats for some forest edge species.

Pine-grassland species

Occurrence and abundance of pine-grassland bird species will vary with changes in understory. In addition to red-cockaded woodpecker, we found brown-headed nuthatch, Bachman's sparrow, chipping sparrow, and red-headed woodpecker are strongly associated with habitat attributes of pine-grasslands. These species favor open stands characterized by herbaceous understory and sparse shrubs. Pine warbler and eastern wood pewee also prefer open midstory, but may be more tolerant of overstory hardwoods. Prescribed burning at periodic intervals (3-5 years) will maintain forest structure required by these species. As woody shrub layer increases with time postburn, indigo bunting and prairie warbler will increase. Fire regimes that eliminate woody shrubs will exclude these two species.

Small Mammal Community

Pine-grassland restoration enhances the entire small mammal community in terms of abundance, richness and diversity. Without exception no species was disadvantaged by restoration treatments. WSI and WSI and fire created a more structurally complex understory than in untreated stands. This was important for habitat generalist species such as the white-footed mouse but also for habitat specialist species such as the golden mouse, harvest mouse, and the hispid cotton rat. The later two species are notable because neither occurred in control stands (Figure 4; Masters et al. 1998). Any treatment that reduces overstory and allows for an increase in understory production will likely benefit this range of species. Fire likely played an important role here as it reduced the stature (and thus shading effect) of woody plants and fire reduced the litter layer, promoting increases in herbaceous production (Masters et al. 1998). Here again, the brief succession windows of suitable habitat created by periodic fire were important for some of the small mammal species (Figure 4).

Management Implications

Pine-grassland communities undoubtedly existed across a broad continuum of canopy closures and basal areas and should be viewed within a landscape-level context. The landscape mosaic was likely composed of a range of communities from open pine woodlands to more dense hardwood-dominated sites such as mesic north slopes and drainages (Foti and Glenn 1991). Breeding bird habitat relationships reveal a need to provide variability in stand structure and composition. This variability should also enhance the range of small mammal species in these stands. For both vertebrate communities, the importance of periodic fire in creating brief windows of suitable understory and midstory habitat structure for some habitat specialist species cannot be overemphasized. Upper and lower ends of targeted stem densities and basal areas may be equally important in maintaining representative habitats across the landscape. Restoration of pine-bluestem habitats will be most effective on sites of historical occurrence, including xeric ridges and south slopes. In addition, the current practice of maintaining hardwood corridors along stream courses and drainages that dissect or encompass pine-bluestem stands should be continued to benefit numerous closed canopy and other species. We found

Table 4.—Breeding bird responses (percent frequency of occurrence in bird plots) to wildlife stand improvement (WSI) and prescribed fire in pine-oak stands on the Ouachita National Forest, Arkansas, May 1992, 1993, 1999 and 2000[1]

	Treatment[2]					*P* > *F*		
	CONTROL	WSI	WSIB1	WSIB2	WSIB3	Contrasts[3]		
Group, Species	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Trt	C vs WSI	NB vs B
Forest Interior								
Acadian Flycatcher	4.8(2.3)	8.3(3.8)	2.6(1.3)	2.8(1.4)	3.2(1.3)	0.181	0.876	0.035
American Redstart	0.8(0.8)	6.0(2.9)	0.7(0.7)	1.6(1.1)	0.0(0.0)	0.160	0.528	0.027
Black & White Warbler	32.1(7.0) a	25.0(5.4) a	6.9(3.6) b	12.7(3.4) b	13.1(5.3) b	0.001	0.001	0.003
Broad-winged Hawk	3.2(1.3)	1.6(0.9)	1.0(0.7)	0.4(0.4)	0.8(0.8)	0.095	0.017	0.148
Carolina Chickadee	17.9(5.2)	16.7(4.0)	12.4(3.9)	13.1(4.4)	8.3(2.9)	0.626	0.332	0.282
Cooper's Hawk	0.0(0.0)	0.8(0.8)	0.3(0.3)	0.0(0.0)	0.4(0.4)	0.697	0.414	0.693
Downy Woodpecker	0.8(0.8)	0.4(0.4)	0.3(0.3)	2.0(1.1)	2.0(1.4)	0.754	0.548	0.530
Great-crested Flycatcher	16.7(3.2) b	18.6(4.3) b	28.8(4.2) ab	39.3(5.7) a	18.3(3.4) b	0.022	0.087	0.072
Hairy Woodpecker	2.4(1.4)	4.0(1.6)	3.6(1.5)	1.6(1.1)	2.0(1.1)	0.735	0.924	0.420
Hooded Warbler	0.0(0.0)	0.0(0.0)	0.3(0.3)	0.0(0.0)	0.0(0.0)	0.523	0.672	0.585
Kentucky Warbler[4]	1.2(1.2)	2.4(1.3)	1.6(0.9)	2.4(1.7)	2.4(1.7)	0.954	0.547	0.641
Ovenbird	39.3(9.8) a	1.6(1.6) b	0.0(0.0) b	0.0(0.0) b	0.0(0.0) b	<0.001	<0.001	0.328
Pileated Woodpecker	27.4(3.1)	24.6(4.6)	24.5(3.6)	20.2(3.0)	14.7(2.8)	0.063	0.038	0.435
Red-bellied Woodpecker	0.4(0.4)	2.0(1.1)	3.6(1.3)	1.2(1.2)	4.8(1.9)	0.117	0.124	0.422
Red-eyed Vireo	58.3(7.5)	56.4(4.1)	45.1(5.8)	54.8(5.5)	40.1(6.3)	0.073	0.057	0.128
Ruby-throated Hummingbird	0.4(0.4)	1.2(0.9)	1.6(0.9)	0.4(0.4)	1.6(1.1)	0.934	0.604	0.798
Scarlet Tanager	16.7(5.1)	10.7(2.8)	7.2(1.8)	4.8(3.2)	8.7(4.0)	0.130	0.082	0.174
Summer Tanager	30.6(5.6)	26.6(4.3)	37.6(3.9)	40.1(5.7)	28.6(4.9)	0.228	0.832	0.070
Tufted Titmouse	39.7(6.6)	27.8(4.4)	28.4(4.3)	19.8(7.4) 2	3.0(6.7)	0.115	0.037	0.395
Whip-poor-will[4]	0.8(0.5)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.414	0.061	1.000
Wood Thrush	2.4(2.4)	0.0(0.0)	0.7(0.7)	0.0(0.0)	0.4(0.4)	0.722	0.645	0.454
Worm-eating Warbler	0.0(0.0)	0.0(0.0)	1.0(1.0)	0.0(0.0)	0.0(0.0)	0.523	0.672	0.585
Forest Edge and Other								
American Crow	25.8(4.0)	32.1(5.8)	30.7(5.4)	34.1(5.9)	32.9(6.8)	0.892	0.356	0.915
American Goldfinch	9.9(3.4) b	11.5(3.9) b	22.5(3.6) a	23.0(5.3) a	26.6(6.4) a	0.024	0.020	0.010
American Kestrel	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.4(0.4)	0.402	0.610	0.511
Barred Owl	1.2(0.9)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.063	0.005	1.000
Blue Grosbeak	0.0(0.0)	2.0(1.1)	1.0(0.7)	0.0(0.0)	0.8(0.8)	0.061	0.115	0.021
Blue-gray Gnatcatcher	0.4(0.4) b	8.3(2.7) a	6.9(1.8) a	1.6(0.9) b	3.6(2.8) b	0.003	0.006	0.017
Blue Jay	24.2(5.2)	11.9(3.5)	6.2(2.1)	6.3(1.9)	8.7(3.2)	0.083	0.010	0.302
Brown-headed Cowbird	1.2(0.9) b	17.1(2.7) a	21.6(4.8) a	20.6(3.9) a	11.5(3.3) a	<0.001	<0.001	0.546
Brown Thrasher	1.6(1.6)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.492	0.082	1.000
Catbird	0.0(0.0)	0.4(0.4)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.402	0.610	0.065
Carolina Wren	21.8(4.7) c	79.0(4.7) a	51.6(7.2) b	45.6(6.9) b	46.4(5.6) b	<0.001	0.001	<0.001
Cedar Waxwing	0.0(0.0)	0.0(0.0)	1.2(1.2)	0.9(0.9)	0.0(0.0)	0.700	0.587	0.487
Chimney Swift	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.6(0.6)	0.0(0.0)	0.450	0.627	0.531
Common Flicker	0.8(0.5)	2.0(1.1)	7.5(2.2)	2.8(1.1)	5.2(2.1)	0.202	0.122	0.146
Common Grackle	0.0(0.0)	0.0(0.0)	0.4(0.4)	0.0(0.0)	0.0(0.0)	0.519	0.676	0.589
Common Yellowthroat	0.0(0.0) c	5.2(3.5) bc	4.2(2.3) ab	13.1(5.8) a	17.1(8.2) ab	0.011	0.004	0.068
Eastern Bluebird	0.0(0.0)	0.8(0.5)	3.9(2.6)	0.8(0.8)	0.4(0.4)	0.210	0.116	0.687
Field Sparrow	0.0(0.0)	1.6(1.6)	0.0(0.0)	0.0(0.0)	0.8(0.8)	0.488	0.454	0.324
Fish Crow	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.4(0.4)	0.414	0.615	0.516
Great-horned Owl	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.4(0.4)	0.4(0.4)	0.397	0.410	0.291
Mourning Dove	0.0(0.0)	4.4(1.9)	5.9(2.9)	5.6(3.3)	6.3(3.8)	0.233	0.031	0.764
Northern Cardinal	6.0(2.2)	9.5(3.4)	4.2(2.0)	0.8(0.8)	4.4(3.2)	0.287	0.569	0.062
Northern Parula	0.0(0.0)	0.0(0.0)	0.3(0.3)	0.0(0.0)	0.0(0.0)	0.523	0.672	0.585

Continued

Table 4.—continued

	Treatment[2]					*P > F*		
	CONTROL	WSI	WSIB1	WSIB2	WSIB3		Contrasts[3]	
Group, Species	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Trt	C vs WSI	NB vs B
Purple Martin	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.9(0.9)	0.500	0.644	0.553
Red-shouldered Hawk	2.8(1.5)	0.8(0.8)	3.3(2.3)	0.8(0.8)	0.0(0.0)	0.494	0.142	0.915
Red-tailed Hawk	0.4(0.4)	0.0(0.0)	2.0(1.2)	1.2(0.9)	0.4(0.4)	0.340	0.779	0.085
Screech Owl	0.4(0.4)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.402	0.058	1.000
Turkey Vulture	0.0(0.0)	1.6(1.1)	0.3(0.3)	0.0(0.0)	0.0(0.0)	0.493	0.553	0.104
White-breasted Nuthatch	2.0(1.1) b	19.0(4.5) a	20.6(2.8) a	25.4(4.9) a	15.9(4.0) a	0.024	0.003	0.664
White-eyed Vireo	0.0(0.0)	2.8(1.5)	0.0(0.0)	1.6(1.1)	0.0(0.0)	0.127	0.274	0.043
Wild Turkey	2.4(2.4)	0.8(0.8)	4.2(2.7)	7.9(2.5)	4.0(1.8)	0.100	0.152	0.055
Wood Duck	0.0(0.0)	0.0(0.0)	0.6(0.6)	0.9(0.9)	0.0(0.0)	0.673	0.587	0.488
Yellow-billed Cuckoo	17.1(4.8)	18.3(4.9)	7.8(2.3)	13.9(4.9)	14.7(4.5)	0.352	0.428	0.205
Yellow-breasted Chat	3.2(1.8) b	29.4(8.0) a	10.1(2.9) b	34.1(8.9) a	43.3(8.1) a	<0.001	<0.001	0.882
Yellow-throated Vireo	0.4(0.4)	6.0(3.2)	8.5(2.7)	4.8(2.2)	6.7(2.7)	0.342	0.092	0.543
Pine-grassland								
Bachman's Sparrow[4]	0.0(0.0) c	2.0(2.0) c	4.6(2.9) bc	11.1(4.0) ab	13.5(4.8) a	0.019	0.023	0.037
Brown-headed Nuthatch[4]	0.0(0.0) c	0.8(0.8) bc	7.8(3.3) ab	8.7(3.7) a	6.3(2.3) ab	0.030	0.025	0.017
Chipping Sparrow	3.2(1.8) c	36.5(8.0) b	69.0(4.9) a	55.6(6.5) ab	55.6(6.9) ab	<0.001	<0.001	0.010
Eastern Wood Pewee	0.8(0.8) c	38.5(5.1) b	53.3(4.0) a	40.1(4.3) ab	32.9(6.5) b	<0.001	<0.001	0.371
Indigo Bunting	19.4(5.3) b	73.4(4.3) a	69.6(4.9) a	78.2(4.3) a	77.0(5.0) a	<0.001	<0.001	0.582
Northern Bobwhite	2.8(1.7) c	34.1(6.7) a	14.4(3.7) b	15.5(6.4) bc	32.1(9.9) ab	0.011	0.005	0.017
Pine Warbler	94.8(1.7)	92.5(2.6)	95.8(1.9)	96.0(1.4) 9	5.2(1.9)	0.504	0.745	0.144
Prairie Warbler[4]	1.2(0.9) c	27.8(7.1) ab	13.4(4.1) bc	39.3(8.0) a	48.0(8.6) a	0.002	<0.001	0.291
Red-cockaded Woodpecker[4]	0.0(0.0) b	0.0(0.0) b	3.6(1.7) a	2.4(1.1) a	3.2(1.8) a	0.014	0.020	0.005
Red-headed Woodpecker[4]	0.0(0.0) b	7.5(4.1) ab	14.7(3.3) a	9.1(4.2) ab	5.6(2.7) ab	0.045	0.015	0.335

[1] Column means within rows followed by the same letter or without letters were not significantly different at the 0.05 level (LSD).

[2] Control=no treatment; WSI= midstory removal and no burning of downed logging slash or trees; WSIB1=midstory removal, late-winter prescribed burn, following first growing season after fire; WSIB2=midstory removal, late-winter prescribed burn, following first growing season after fire; WSIB3=midstory removal, late-winter prescribed burn, following first growing season after fire.

[3] Specific orthogonal contrasts: C=control, WSI=midstory removal, NB=nonburned WSI stands, B=WSI stands burned within previous 3 years.

[4] Priority species (Pashley et al. 2000).

Carolina wren, Kentucky warbler, red-eyed vireo and tufted titmouse were closely associated with greenbelts or drainages dominated by hardwoods extending into the interior of treated stands. Residual hardwoods retained within the stand were important to red-eyed vireo, summer tanager, scarlet tanager, tufted titmouse, yellow-throated vireo and white-breasted nuthatch occurrence within a restored stand.

Although relative abundance and frequency of occurrence increased for many species, we recognize that this does not necessarily translate to high reproductive success. Nest predation and parasitism can be a major concern in pine forests of the southeast U.S. (White et al. 1999). A concern is raised here because of increased brown-headed cowbird abundance in stands that had been recently thinned (WSI) and in the first and second growing season following fire (WSIB1 and WSIB2 treatments respectively). We did observe fledglings during point-counts of brown-headed nuthatch, Carolina wren, chipping sparrow, common yellowthroat, indigo bunting, ovenbird and pine warbler in stands associated with their highest densities, but not all observers noted occurrence of fledglings. The influence of cowbird nest parasitism in restored stands should be further examined.

While knowledge of individual species relationships to habitat components is vital to manage for habitat availability (Wilson et al. 1995, Masters et al. 1998), regional biodiversity will likely be maximized by restoring the natural disturbance regime (Vose 2000). By mimicking fire events of the past, the most complete range of habitats and attendant species may best be maintained over time in the Ouachita Mountains ecosystem and possibly shortleaf pine dominated systems elsewhere. With repeated alternating shifts from frequent burning (≤3 year intervals) to no burning and periodic shifts to growing season burns, current stands may eventually approach structural conditions reported historically as they are repeatedly thinned. Irregular burn intervals may be particularly important to ensure adequate pine recruitment (Cain 1993; Masters et al 1995). A return to the presettlement fire regime will not eliminate residual hardwoods because large hardwoods are tolerant even of frequent fire (Waldrop et al. 1992). Plant and animal response to large-scale burns and growing season burns should be investigated as such burns were probably characteristic of the original fire regime (Masters et al 1995).

Landscape-level studies are necessary to determine avian relationships to stand sizes and juxtaposition of pine-bluestem woodlands and pine-oak forest. Geographic information systems and ecological models should be used to ensure the presence of uncommon natural communities across the landscape and into the future. We recommend additional research on red-cockaded woodpecker population viability in low-density stands with varying proportions of hardwoods because overall stand density may be as critical as some upper level threshold for a maximum hardwood component. Research is also needed to determine the extent and importance of other fire-dependent community types, including oak-dominated woodlands and savannas and breeding bird and small mammal habitat relationships within those settings.

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List of Common and Scientific Names of all Occurring Breeding Bird and Small Mammal Species

Common Name	Scientific Name (authority)
Birds	
Acadian Flycatcher	*Empidonax virescens (Vieillot)*
American Crow	*Corvus brachyrhynchos (Brehm)*
American Goldfinch	*Carduelis tristis (Linnaeus)*
American Kestrel	*Falco sparverius (Linnaeus)*
American Redstart	*Setophaga ruticilla* (Linnaeus)
Bachman's Sparrow	*Aimophila aestivalis (Lichtenstein)*
Barred Owl	*Strix varia* (Barton)
Black and White Warbler	*Mniotilta varia* (Linnaeus)
Blue Grosbeak	*Guiraca caerulea* (Linnaeus)
Blue Jay	*Cyanocitta cristata* (Linnaeus)
Blue-gray Gnatcatcher	*Polioptila caerulea (Linnaeus)*
Broad-winged Hawk	*Buteo platypterus (Vieillot)*
Brown Thrasher	*Toxostoma rufum (Linnaeus)*
Brown-headed Cowbird	*Molothrus ater (Boddaert)*
Brown-headed Nuthatch	*Sitta pusilla (Latham)*
Carolina Chickadee	*Poecile carolinensis (Audubon)*
Carolina Wren	*Thryothorus ludovicianus (Latham)*
Catbird	*Dumetella carolinensis (Linnaeus)*
Cedar Waxwing	*Bombycilla cedrorum (Vieillot)*
Chimney Swift	*Chaetura pelagica (Linnaeus)*
Chipping Sparrow	*Spizella passerina (Bechstein)*
Common Flicker	*Colaptes auratus (Linnaeus)*
Common Grackle	*Quiscalus quiscula (Linnaeus)*
Common Yellowthroat	*Geothlypis trichas (Linnaeus)*
Cooper's Hawk	*Geothlypis trichas (Bonaparte)*
Downy Woodpecker	*Picoides pubescens (Linnaeus)*
Eastern Bluebird	*Sialia sialis (Linnaeus)*
Eastern Wood Pewee	*Contopus virens (Linnaeus)*
Field Sparrow	*Spizella pusilla (Wilson)*
Fish Crow	*Corvus ossifragus (Wilson)*
Great-crested Flycatcher	*Myiarchus crinitus (Linnaeus)*
Great-horned Owl	*Bubo virginianus (Gmelin)*
Hairy Woodpecker	*Picoides villosus (Linnaeus)*
Hooded Warbler	*Wilsonia citrina (Boddaert)*
Indigo Bunting	*Passerina cyanea (Linnaeus)*
Kentucky Warbler	*Oporornis formosus (Wilson)*
Mourning Dove	*Zenaida macroura (Linnaeus)*
Northern Bobwhite	*Colinus virginianus* (Linnaeus)
Northern Cardinal	*Cardinalis cardinalis (Linnaeus)*
Northern Parula	*Parula americana (Linnaeus)*
Ovenbird	*Seiurus aurocapillus (Linnaeus)*
Pileated Woodpecker	*Dryocopus pileatus (Linnaeus)*
Pine Warbler	*Dendroica pinus (Wilson)*
Prairie Warbler	*Dendroica discolor (Vieillot)*
Purple Martin	*Progne subis (Linnaeus)*
Red-bellied Woodpecker	*Melanerpes carolinus (Linnaeus)*
Red-cockaded Woodpecker	*Picoides borealis* (Vieillot)

continued

List of Common and Scientific Names—continued

Common Name	Scientific Name (authority)
Red-eyed Vireo	*Vireo olivaceus (Linnaeus)*
Red-headed Woodpecker	*Melanerpes erythrocephalus (Linnaeus)*
Red-shouldered Hawk	*Buteo lineatus (Gmelin)*
Red-tailed Hawk	*Buteo jamaicensis (Gmelin)*
Ruby-throated Hummingbird	*Archilochus colubris (Linnaeus)*
Scarlet Tanager	*Piranga olivacea (Gmelin)*
Screech Owl	*Otus asio (Linnaeus)*
Summer Tanager	*Piranga rubra (Linnaeus)*
Tufted Titmouse	*Baeolophus bicolor (Linnaeus)*
Turkey Vulture	*Cathartes aura* (Linnaeus)
Whip-poor-will	*Caprimulgus vociferous (Wilson)*
White-breasted Nuthatch	*Sitta carolinensis (Latham)*
White-eyed Vireo	*Vireo griseus (Boddaert)*
Wild Turkey	*Meleagris gallopavo* (Linnaeus)
Wood Duck	*Aix sponsa (Linnaeus)*
Wood Thrush	*Hylocichla mustelina (Gmelin)*
Worm-eating Warbler	*Helmitheros vermivorus (Gmelin)*
Yellow-billed Cuckoo	*Coccyzus americanus (Linnaeus)*
Yellow-breasted Chat	*Icteria virens (Linnaeus)*
Yellow-throated Vireo	*Vireo flavifrons (Vieillot)*
Small Mammals	
Cotton Mouse	*Peromyscus gossypinus* (LeConte)
Deer Mouse	*Peromyscus maniculatus* (Wagner)
Eastern Woodrat	*Neotoma floridana* (Ord)
Fulvous Harvest Mouse	*Reithrodontomys fulvescens* (J.A. Allen)
Hispid Cotton Rat	*Sigmodon hispidus* (Say and Ord)
Golden Mouse	*Ochrotomys nutalli* (Harlan)
Pine Vole	*Microtus pinetorum* (LeConte)
Southern Short-tailed Shrew	*Blarina carolinensis* (Bachman)
White-footed Mouse	*Peromyscus leucopus* (Rafinesque)

Fire, Habitat Structure and Herpetofauna in the Southeast

Cathryn H. Greenberg[1]

Abstract.—In this paper I review studies of herpetofauna in two fire-maintained, xeric pineland southeastern U.S. ecosystems, longleaf pine-turkey oak sandhills and Florida sand pine scrub. I address evolutionary adaptations of herpetofauna to these xeric environments, and how fire disturbance influences herpetofaunal community composition by structuring habitat. Where data are available I examine how anthropogenic disturbance such as fire suppression, clearcutting, and restoration treatments can affect herpetofaunal community composition, and whether some anthropogenic disturbance types can mimic natural disturbance in their effects. I also address possible reasons why detection of population response is more difficult for amphibians than for common reptile species.

Natural disturbance is an important force in shaping the habitat structure of ecosystems within southeastern United States. However, type, frequency, intensity, and scale of natural disturbance varies among ecosystems. Wind, ice, insect and disease outbreaks, or fire can be dominant disturbance types in some southeastern ecosystems, acting at scales ranging from single-tree death to thousands of hectares. Disturbance regimes and the habitat structure they maintain exert selective pressure on life history strategies, physical, and behavioral adaptations of resident flora and fauna (Denslow 1980). Hence, biotic components that characterize and distinguish ecosystems often provide clues to the most common habitat structure, and dominant disturbance regime that historically shaped it.

Interference with disturbance regimes can alter the habitat structure of ecosystems by disrupting processes that shape, maintain, and perpetuate them. Major structural alterations to habitat can, in turn, potentially adversely affect characteristic flora and fauna. Similarly, introduction of anthropogenic disturbance types to ecosystems that do not resemble or mimic the results of natural disturbance regimes can adversely affect their biota.

Fire is an integral disturbance type to many southeastern ecosystems that drives their process, function, and many of their defining floral and faunal components (Myers and Van Lear 1998). However, fire characteristics vary among ecosystems. In the southeastern Coastal Plain, the longleaf pine-wiregrass ecosystem was historically maintained by low-intensity, high frequency (3-5 year intervals), large-scale groundfires that killed many young, invading hardwoods. This burning regime maintained a low density of mature longleaf pine, scattered understory oaks, and an abundance of sunlight at ground level that promoted a continuous groundcover dominated by wiregrass and a high diversity of herbaceous plants (Myers 1990). Fire suppression is more aptly viewed as a disturbance in sandhills than is fire itself, as it disrupts a process that maintains and perpetuates this habitat structure. In the absence of fire a dense hardwood understory develops, leaf litter and shade increase, and wiregrass becomes sparse (see Table 1).

Conversely, the sand pine scrub ecosystem in peninsular Florida was maintained by high-intensity, low frequency (10-100 year intervals) wildfires that killed virtually all above-ground vegetation, followed by rapid resprouting of shrubs (Abrahamson 1984; Greenberg et al. 1995). During long fire-free intervals a thick shrub layer developed, and even-aged stands of sand pine matured at densities that varied dramatically among scrubs.

Herpetofaunal assemblages differ among southeastern ecosystems and physiographical regions according to historical biogeography, and physiological and environmental constraints that adapt each species to specific or generalized habitats. Species exhibiting specialized traits often are also restricted in their local distribution by habitat type and condition. Reptile and amphibian species in xeric sandhills and scrub include widespread generalists and species having narrow habitat requirements, usually typical of recent post-fire conditions. Because the habitat condition in xeric sandhills and sand pine scrub is sculpted by fire, herpetofaunal distribution and abundance is closely tied to disturbance regime.

Surprisingly few studies examine differences in habitat characteristics between fire-suppressed and fire-maintained sandhills or sand pine scrub, and how habitat structure affects herpetofauna. However, even an intuitive understanding of fire effects and ecological requirements of herpetofauna would suggest that there are impacts. Species having specialized adaptations to structural features of fire-maintained habitat, such as bare ground, could be at a disadvantage in fire suppressed habitat. Gopher tortoises are abundant in regularly burned sandhills, where a dense layer of wiregrass and herbaceous groundcover provide plentiful food, and the light level is adequate for nest sites (Diemer 1992). This might be expected to adversely affect gopher tortoise burrow commensals, but only if burrows are a limiting resource. By reducing root density in uplands (by reducing tree density), fire likely

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[1] Southern Research Station, USDA Forest Service, Asheville, NC 28806.

Table 1.—Mean (±SE) percent cover of select habitat characteristics and basal area (BA) of trees in hardwood-invaded (HI) versus savanna-like (SL) longleaf pine-wiregrass sandhills surrounding eight isolated, ephemeral ponds in the Ocala National Forest, Florida. Percentage data were square root-arcsine transformed for t-tests but are presented as untransformed values (adapted from Greenberg in press)

	Upland matrix			
	HI	SL	df	P
Wiregrass (% cover)	11.6± 6.0	66.5±8.6	25.0	<0.0001
Herbaceous (% cover)	4.5 ± 1.4	21.5±4.7	25.0	0.0002
Leaf litter (% cover)	99.3± 0.5	68.5±9.6	12.7	0.0023
Bare ground (% cover)	0.4 ± 0.2	9.4±3.7	13.3	0.0072
Coarse woody debris ≥12.5 cm (% cover)	0.7 ± 0.4	1.2±0.6	25.0	0.6208
Shrub <2.5 cm (% cover)	47.5±8.2	27.8±6.4	25.0	0.0548
Light (%)	22.8±3.7	54.0±4.9	25.0	<0.0001
Longleaf pine BA (m^2)	4.5 ±1.2	4.8±1.1	25.0	0.8475
Hardwoods & sand pine BA (m^2)	2.8±0.7	0.9±0.5	25.0	0.0357
Snag BA (m^2)	0.7±0.3	0.5±0.3	25.0	0.5677

facilitates burrowing conditions for primary underground excavators (such as gopher tortoises and pocket gophers) and thereby secondary burrow- and tunnel users such as Florida gopher frogs and striped newts. Historically, fire probably maintained and deepened ephemeral wetlands that are used as breeding sites by xeric pineland amphibians by killing encroaching trees and shrubs, and burning peat and vegetation buildup during dry periods. Fire also maintained a low density of mature trees in uplands surrounding ponds by killing young trees. Lower tree density reduces transpiration, but may increase evaporation from ponds. Hardwood invasion may affect hydroperiod and water depth of shallow ponds by altering the balance between transpiration and evaporation (Sharitz and Gresham 1998).

In this paper I will review studies of herpetofauna in two fire-maintained xeric pineland ecosystems, longleaf pine-turkey oak-wiregrass sandhills and Florida sand pine scrub. I will specifically address some evolutionary adaptations of herpetofauna to xeric environments of these fire-maintained ecosystems, and how disturbance influences herpetofaunal community composition by structuring habitat. I also will discuss possible reasons why detection of population response is more difficult for amphibians than for common reptile species. Where data are available I examine how anthropogenic disturbance such as fire suppression, clearcutting, and restoration treatments can affect herpetofaunal community composition, and whether some anthropogenic disturbance types can mimic natural disturbance in their effects.

Although reptiles and amphibians often are generically lumped together as "herpetofauna," they are phylogenetically as distinct from one another as are mammals from birds. Amphibians (class *Amphibia*) have permeable, moist skin that for many serves a respiratory function, and increases their susceptibility to desiccation. Many species also require water for egg deposition and larval development. Taxa vary considerably in their vulnerability to desiccation. For example, anurans are more tolerant of high temperatures (Stebbins and Cohen 1995), and can store and reabsorb larger amounts of water in their bladders than salamanders (Zug 1993). Some salamanders are lungless, and some are completely terrestrial (DeMaynadier and Hunter 1995). Many amphibian species have small home ranges (Duellman and Trueb 1986) and poor dispersal capabilities (Sinsch 1990). Conversely, most reptiles (class *Reptilia*) require warm temperatures (associated with higher light levels) for egg incubation and successful development of hatchlings (Deeming and Ferguson 1991). Reptiles have dry, scaly skin that protects them from desiccation. Clearly, response to disturbance might be expected to differ between the two classes, and among species within them.

Behavioral or physical adaptations to xeric upland environments likely reduce the vulnerability of the native herpetofauna to fire and consequent habitat alterations (Means and Campbell 1981). Among reptilian sandhill and sand pine scrub inhabitants, several species exhibit specialized behavioral or anatomical traits for surviving in a xeric, open, loose-sand environment (Campbell and Christman 1982a;

Table 2.—Mean (± SE) number of select reptile species captured using drift fences and pitfall traps during August 1991 - September 1992 in three treatments and mature forest controls (n = 3 per treatment and control) in sand pine scrub, Ocala National Forest, Florida

Species	Burn & Salvage	Clearcut & Rollerchop	Clearcut & Bracke-Seed	Mature Forest	P
Six-lined Racerunner	$5.7 \pm 1.2^{a,b}$	12.3 ± 1.2^{a}	13.0 ± 5.2^{a}	2.3 ± 1.5^{b}	0.0752
Mole Skink	11.0 ± 2.1^{a}	16.7 ± 3.5^{a}	9.7 ± 3.5^{a}	1.3 ± 0.3^{b}	0.0237
Southeastern Five-lined Skink	1.3 ± 1.3^{a}	1.0 ± 0.6^{a}	1.0 ± 0.6^{a}	13.3 ± 1.7^{b}	0.0001
Ground Skink	$2.0 \pm 1.0^{a,b}$	0.7 ± 0.7^{a}	0.0 ± 0.0	3.7 ± 0.3^{b}	0.0140
Scrub Lizard	$12.0 \pm 6.5^{a,b}$	$25.0 \pm 11.0^{a,c}$	38.3 ± 2.6^{c}	1.7 ± 0.3^{b}	0.0194
Peninsula crown snake	2.0 ± 1.2^{a}	$7.7 \pm 2.4^{a,b}$	8.7 ± 1.3^{b}	3.7 ± 2.2^{a}	0.0977

Greenberg et al. 1994; Stout et al. 1998). For example, sand skinks, mole skinks, and crown snakes are "sand swimmers," requiring loose, bare sand for semifossorial movement. Short-tailed snakes (Campbell 1992) and pine snakes (Conant and Collins 1991) use underground burrows extensively. Six-lined racerunners and scrub lizards are cursorial, and the mechanics of movement are apparently facilitated by open, bare sand.

Behaviors that exploit cool, moist microhabitats also are apparent in many reptile species inhabiting sandhills and scrub. Gopher tortoises dig burrows that are used as retreats by at least 23 reptile species, including pine snakes and sand skinks, and 9 amphibian species, including gopher frogs and narrowmouth toads (Jackson and Milstrey 1989). Several species, including southern, eastern narrowmouth, and spadefoot toads avoid heat and desiccation by burrowing underground, or by using pre-existing burrows, tunnels and root channels (Russell et al. 1999). Nocturnal foraging, that may allow some species to tolerate extreme conditions of temperature and drought. Selective pressure "pre-adapts" many species to their environments and the disturbance regimes that shape them.

Many reptile species appear to respond to fire-maintained habitat features such as the amount of bare ground, rather than to plant associations *per se* (Campbell and Christman 1982; Greenberg et al. 1994). In sand pine scrub, scrub lizards, sand skinks, six-lined racerunners, mole skinks, and crown snakes are more abundant in open scrub habitat with a high proportion of bare sand than in mature, forested scrub (Campbell and Christman 1982; Greenberg et al. 1994; Means and Campbell 1982). Scrub lizards ($r^2 = 0.44$; $p = 0.0182$) and six-lined racerunners ($r^2 = 68$; $p = 0.0010$) are positively correlated, whereas southeastern five-lined skinks are negatively correlated ($r^2 = 0.75$; $p = 0.0003$) to the amount of bare sand (Greenberg et al.1994).

Reptile response to anthropogenic and natural disturbances may be similar in some ecosystems if the resulting post-disturbance habitat structure is similar. For example, the relative abundance of mole skinks, scrub lizards, and six-lined racerunners was similar in sites that had been recently burned and salvage-logged, clearcut with low-intensity site preparation, and clearcut followed by roller-chopping, but lower in mature forested sand pine scrub (Greenberg et al. 1994). Conversely, generalist species that are common in many southeastern ecosystems, such as ground skinks and southeastern five-lined skinks were widespread among scrub age classes or more abundant in mature sand pine forest than in young, open scrub (Table 2) (Greenberg et al. 1994). This suggests that high-intensity natural disturbance was an important selective pressure on physical or behavioral adaptations by herpetofauna.

There is a dearth of information on how fire suppression affects herpetofauna of xeric sandhills. Potential effects appear difficult to detect, perhaps in part because habitat heterogeneity remains within hardwood-invaded sandhills for decades after commencement of fire suppression. Litt (1999) reported no difference in species richness, and few differences in relative abundance of species between fire-suppressed sandhills and three restoration treatments (mechanical girdling and felling; herbicide; and growing season fire) or frequently burned "reference" sandhills. Meshaka and Layne (pers. comm.) also found that during 1979-1994, species diversity and richness of herpetofauna in a fire-suppressed sandhill in Florida was similar to that reported for frequently burned Florida sandhills. They attributed the persistence of xeric-adapted reptiles such as six-lined racerunners and scrub lizards species to a frequent occurrence of shrub-free openings. The abundance of southeastern five-lined skinks increased during their study. Mushinsky (1985) reported higher abundance of six-lined racerunners on an annually burned sandhill site with low wiregrass and herbaceous cover relative to sites with longer burn intervals. Sandhill reptiles respond to subsets of microhabitat characteristics that are unique to each species or species group (Litt 1999; Mushinsky 1985). Habitat heterogeneity within regularly burned sandhills permits the co-occurrence of post-disturbance specialists and generalists, and contributes to the high diversity of herpetofauna in sandhills and scrub relative to other Florida ecosystems (Campbell and Christman 1982).

Among the few studies comparing amphibian use of fire suppressed versus regularly burned xeric pinelands, none provide conclusive results. Clearly, amphibians that are characteristic of xeric, fire-climax pinelands also use fire-suppressed habitat (Greenberg 2001; Greenberg unpubl. data; Litt 1999; Mushinsky 1985). Palis (1998) reported high adult usage and egg mass deposition by gopher frogs at a pond within fire-suppressed longleaf pine-turkey oak sandhill uplands. Some studies provide anecdotal evidence of habitat preference. Palis (1998) reported a tendency for adult gopher frogs to immigrate to a breeding pond from the direction of an early successional bombing range versus a hardwood-invaded sandhill, suggesting heavier use of the more open habitat. Dodd (1996) reported more captures in open xeric hammock, fewer in closed xeric hammock, and as expected in sandhills if captures were proportional to trap effort.

Season of burn influences diversity patterns, percent cover, and flowering of several herbaceous sandhill species (Robins and Myers 1989). Few studies address whether season of burn affects herpetofaunal communities in sandhills. C.L. Hardy and associates (pers. comm.) applied dormant season burns to uplands surrounding one half of ephemeral ponds in sandhills, and growing season burns to the other half. Their preliminary results indicate no treatment differences in amphibian or reptile use of ponds.

Amphibian population response to disturbance or habitat condition appears to be difficult to detect in fire-adapted ecosystems such as sandhills and sand pine scrub, and may require long-term data. Although many species use uplands during much of their lives, the distribution and abundance of amphibians in uplands is closely tied to distance from water (Dodd 1995, 1996). Greenberg (1993) reported that the relative abundance of anurans was not significantly related to stand age or disturbance treatment in sand pine scrub, but was negatively correlated with distance from permanent water sources (lakes and wet prairies) ($r^2 = 0.5406$; $p = 0.0064$). This relationship was not consistent among individual species, possibly because some small, ephemeral wetlands were not detected, and because some species travel further from breeding sites than others (Dodd 1996). Dodd (1996) reported no significant correlation between the number of amphibians captured per trap and trap distance from nearest water body, but 83% of amphibians were captured within 600-m from water. Greenberg (1993) collected spadefoot toad tadpoles from a roadside puddle in sand pine scrub. This suggests larval developmental rates, precipitation, and the distribution of depressions as small as puddles (in this case, anthropogenically-created by clay-subsidized sand roads) can differentially affect the spatial and temporal distribution patterns of amphibians. Clearly, ephemeral ponds are critical centers of herpetofaunal diversity in sandhills (Dodd 1992; Moler and Franz 1987); most amphibian and several reptile species, such as turtles and swamp snakes, would not occur there in their absence.

Wetlands within sandhills function as a magnet to amphibians. Capture rates may reflect life history patterns such as longevity and strong breeding site fidelity, providing misleading representation of relative abundance among species, and sites, and years. Hydroperiod, water depth, and underwater dynamics of competition and predation also heavily influence annual recruitment and population fluctuations (Heyer 1976; Maiorana 1976), and further confound data interpretation.

As an example, preliminary results of a study comparing herpetofaunal use of isolated, ephemeral ponds using intermittent drift fences with pitfall and funnel traps, indicate significant differences ($p < 0.05$) in the use of ponds within regularly burned sandhills ($n = 4$) versus hardwood-invaded sandhills ($n = 4$) by some amphibian species. Whereas significantly more adult eastern narrowmouth toads, leopard frogs, and southern toads used ponds within hardwood-invaded sandhills during 1994-1997, more adult striped newts used ponds in savanna-like sandhills (Table 3). Juvenile recruitment by oak toads and narrowmouth toads was higher in hardwood-invaded sandhills (Greenberg, unpubl. data), and recruitment by Florida gopher frogs was higher in savanna-like sandhills (Greenberg 2001) (Table 3). However, confidence in the biological significance of these results is compromised by a lack of clean distinction between upland matrix treatments. The level of hardwood invasion in the hardwood-invaded upland matrix is patchy (Table 1). Further, the distance (on one side) between 3 ponds in the hardwood-invaded upland matrix and the savanna-like upland matrix is < 30 m. Variation in pond use and recruitment among species, sites, and years is high. These and confounding factors associated with using ponds as trap foci suggests that detection of biologically meaningful differences in pond use among different habitat conditions requires long-term data and heavy replication.

In recent decades, recognition that fire is critical to xeric pineland ecosystems has led to implementation of control burning programs for sandhill on many public lands, where resources permit. In sandhills, the restoration goal commonly is to create a "park-like" environment dominated by longleaf pine and wiregrass, and virtually devoid of a hardwood midstory. This vision is inspired by some descriptions by early surveyors and naturalists (e.g., Vignoles 1823; Williams 1837; Harper 1911; Myers 1990), and by the narrow

Table 3.—Mean (+ SE) adult use (entering and exiting individuals) and juvenile recruitment (exiting individuals only) of select pond-breeding amphibians using isolated, ephemeral ponds in hardwood-invaded (HI) (n = 4) and savanna-like (SL) (n=4) longleaf pine-turkey oak-wiregrass sandhills during February 1994 - January 1998, Ocala National Forest, Florida

		Adults (No./100 trapnights)			Juveniles (No./100 trapnights)		
		Treatment			Treatment		
Species	Year	HI	SL	$\underline{P} < 0.05$	HI	SL	$\underline{P} < 0.05$
Oak Toad	1	0.61 + 0.1	0.12 + 0.08		0.41 + 0.13	0.01 + 0.00	*
	2	0.19 + 0.09	0.09 + 0.05		0.00 + 0.00	0.00 + 0.00	
	3	0.52 + 0.51	0.11 + 0.06		0.02 + 0.02	0.00 + 0.00	
	4	0.01 + 0.00	0.08 + 0.04		0.00 + 0.00	0.00 + 0.00	
	5	0.77 + 0.57	0.25 + 0.18		0.12 + 0.12	0.00 + 0.00	
Southern Toad	1	0.17 + 0.07	0.03 + 0.01	*	0.00 + 0.00	0.00 + 0.00	
	2	0.21 + 0.03	0.09 + 0.04		0.00 + 0.00	0.00 + 0.00	
	3	0.17 + 0.06	0.10 + 0.02		0.15 + 0.14	0.04 + 0.04	
	4	0.09 + 0.01	0.08 + 0.04		0.00 + 0.00	0.00 + 0.00	
	5	0.20 + 0.11	0.10 + 0.02		2.82 + 2.79	0.02 + 0.02	
Eastern Narrowmouth Toad	1	1.27 + 0.30	0.50 + 0.25	*	0.16 + 0.07	0.01 + 0.01	*
	2	0.47 + 0.16	0.27 + 0.09		0.01 + 0.01	0.00 + 0.00	
	3	0.25 + 0.11	0.19 + 0.07		0.01 + 0.00	0.00 + 0.00	
	4	0.13 + 0.03	0.18 + 0.09		0.01 + 0.01	0.00 + 0.00	
	5	0.13 + 0.03	0.07 + 0.04		0.01 + 0.01	0.00 + 0.00	
Striped Newt	1	0.00 + 0.00	0.01 + 0.01	*	0.00 + 0.00	0.01 + 0.01	
	2	0.00 + 0.00	0.00 + 0.00		0.00 + 0.00	0.00 + 0.00	
	3	0.00 + 0.00	0.00 + 0.00		0.00 + 0.00	0.01 + 0.01	
	4	0.00 + 0.00	0.10 + 0.08		0.00 + 0.00	0.01 + 0.01	
	5	0.05 + 0.04	0.26 + 0.23		0.06 + 0.03	1.23 + 1.19	
Florida gopher frog	1	0.04 + 0.01	0.02 + 0.01		0.00 + 0.00	0.01 + 0.00	*
	2	0.01 + 0.00	0.02 + 0.01		0.15 + 0.04	0.60 + 0.27	
	3	0.01 + 0.00	0.02 + 0.01		0.07 + 0.03	0.33 + 0.14	
	4	0.00 + 0.00	0.01 + 0.00		0.01 + 0.01	0.06 + 0.02	
	5	0.01 + 0.00	0.01 + 0.01		0.32 + 0.04	0.72 + 0.27	
Bullfrog	1	0.00 + 0.00	0.01 + 0.00		0.02 + 0.01	0.06 + 0.04	
	2	0.00 + 0.00	0.00 + 0.00		0.01 + 0.01	0.01 + 0.00	
	3	0.00 + 0.00	0.00 + 0.00		0.02 + 0.01	0.03 + 0.00	
	4	0.00 + 0.00	0.00 + 0.00		0.01 + 0.01	0.01 + 0.01	
	5	0.00 + 0.00	0.00 + 0.00		0.10 + 0.06	0.33 + 0.24	
Pig Frog	1	0.00 + 0.00	0.00 + 0.00		0.00 + 0.00	0.00 + 0.00	
	2	0.00 + 0.00	0.00 + 0.00		0.03 + 0.02	0.04 + 0.02	
	3	0.01 + 0.00	0.00 + 0.00		0.02 + 0.01	0.07 + 0.03	
	4	0.00 + 0.00	0.00 + 0.00		0.03 + 0.02	0.15 + 0.11	
	5	0.00 + 0.00	0.00 + 0.00		0.05 + 0.02	0.03 + 0.02	
Leopard Frog	1	0.09 + 0.02	0.07 + 0.04	*	0.25 + 0.13	0.55 + 0.50	
	2	0.06 + 0.02	0.04 + 0.01		0.11 + 0.03	0.14 + 0.07	
	3	0.04 + 0.01	0.03 + 0.00		0.12 + 0.04	0.11 + 0.03	
	4	0.01 + 0.00	0.01 + 0.00		0.01 + 0.01	0.01 + 0.01	
	5	0.03 + 0.01	0.01 + 0.01		0.40 + 0.10	0.17 + 0.11	
Spadfoot Toad	1	0.31 + 0.21	0.73 + 0.47		10.70 + 10.68	0.00 + 0.00	
	2	0.37 + 0.34	0.08 + 0.07		4.58 + 4.54	0.00 + 0.00	
	3	0.10 + 0.10	0.02 + 0.01		0.06 + 0.05	0.00 + 0.00	
	4	0.00 + 0.00	0.00 + 0.00		0.00 + 0.00	0.00 + 0.00	
	5	0.19 + 0.17	0.10 + 0.05		0.00 + 0.00	0.00 + 0.00	

habitat requirements of the endangered red-cockaded woodpecker (Carter et al. 1995). However, such descriptions may over-represent this sandhill variant because it was more easily (and commonly) traveled (Landers et al. 1990).

Palynological evidence suggests that the relative dominance of oak and pine has shifted many times within the past 20,000 years in the southeastern United States as a result of climatic and sea level changes that affects soil moisture and fire frequency (Watts 1971). Descriptions by early naturalists (see Myers 1990), plant composition and characteristics, and the presence of old-growth oaks suggest that even within recent centuries, fire in sandhills was spatially and temporally variable (Greenberg and Simons 1999; also see Myers 1990). Prior to fire suppression, patchiness in the intensity, season, frequency, and spatial extent of fire allowed some young oaks to reach fire-resistant size and occur in varying densities over time and across the sandhills landscape (Greenberg and Simons 1999).

Indeed, several lines of evidence suggest that tree-sized oaks were an integral part of the sandhill landscape (Greenberg and Simons 1999). Several oak species, including bluejack, turkey, and sand post oak are endemic to xeric pinelands, suggesting that they historically occurred there. The ability to reach tree size and reproduce sexually (as well as clonally) indicates that trees reached maturity frequently enough to retain the trait (Berg and Hamrick 1994; Landers et al. 1990; Myers 1990). The presence of acorn-dependent, xeric-pineland species such as Sherman's fox squirrels and red-headed woodpeckers suggests that hard mast was produced in sandhills. Clearly, stand conditions within these ecosystems historically were not homogeneous, but a heterogeneous matrix of stand ages and structural conditions.

The presence of both specialized, xeric-adapted species and wide-ranging generalist species in sandhills and scrub further indicate that historically, variable burning intensities, intervals, and spatial extent created a range of microhabitats and stand conditions (Campbell and Christman 1982). Braithwaite (1987) reported that lizard species in the wet-dry tropics of Australia were differentially favored in habitats created by various fire regimes. He suggested that no single fire regime was optimal for all species, but that a range of burning regimes would retain the entire lizard community.

Historically, fire frequency, intensity, and behavior as it interacted with fuel loads, wind, precipitation, topography, and pre-existing vegetation patterns was unlikely to sculpt homogeneous habitat structures across large landscapes. Whereas the scale of heterogeneity in stand age, tree or shrub density, and associated habitat features differed between sandhills (smaller habitat patches) and scrub (larger habitat patches), both ecosystems were a mosaic of stand conditions that shifted spatially and temporally across the landscape (Greenberg et al. 1994). Populations of xeric adapted reptiles probably became locally extinct, or persisted in low numbers within long-unburned habitat patches, exploiting young, open, suitable habitat through colonization and reproduction as it was created by fire.

Ideally, ecosystem management systems should be designed to incorporate silvicultural and land management systems that mimic natural disturbance (Hansen et al., 1991; Greenberg et al., 1994). A necessary corollary is the need to identify habitat characteristics that promote diversity and abundance of species (Hansen et al., 1991). More research on herpetofaunal response to natural disturbance and the associated changes in habitat structure is critically needed in order to gauge the success or failure of management.

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List of Species Names

Vegetation	
Wiregrass	*Aristida stricta* (Micheaux)
Turkey oak	*Quercus laevis* (Walt.)
Bluejack oak	*Quercus incana* (Bartr.)
Sand post oak	*Quercus margaretta* (Ashe)
Longleaf pine	*Pinus palustris* (Mill.)
Sand pine	*Pinus clausa* (Chapm. Ex Engelm.) Vasey ex Sarg.
Birds	
Red-cockaded woodpecker	*Picoides borealis*
Red-headed woodpecker	*Melanerpes erythrocephalus*
Mammals	
Sherman's fox squirrels	*Sciurus niger shermani*
Herpetofauna	
Six-lined racerunner	*Cnemidophorus sexlineatus*
Gopher tortoise	*Gopherus polyphemus*
Short-tailed snake	*Stilosoma extenuatum*
Pine snake	*Pituophis melanoleucus*
Scrub lizard	*Sceloporus woodi*
Sand skink	*Neoseps reynoldsi*
Mole skink	*Eumeces egregius*
Southeastern five-lined skink	*Eumeces inexpectatus*
Ground skink	*Scincella laterale*
Peninsula crown snake	*Tantilla relicta*
Oak toad	*Bufo quercicus*
Southern toad	*Bufo terrestris*
Striped newt	*Notophthalmus perstriatus*
Spadefoot toad	*Scaphiopus holbrookii*
Narrowmouth toad	*Gastrophryne carolinensis*
Florida gopher frog	*Rana capito aesopus*
Pig frog	*Rana grylio*
Bullfrog	*Rana catesbeiana*
Leopard frog	*Rana utricularia*

Numerical Response of Small Vertebrates to Prescribed Fire in California Oak Woodland

Justin K. Vreeland[1] *and William D. Tietje*[2]

Abstract.—Use of prescribed fire is increasing in California oak woodlands, but its effects on vertebrate wildlife are unknown. We conducted a light-intensity prescribed fire in mixed blue oak–coast live-oak woodlands in coastal-central California and assessed vegetation change and numerical response of small, nongame vertebrates to the fire. Four of 13 vegetation and habitat components that we measured were reduced significantly ($P < 0.05$) by the fire. We observed no change in relative abundance of small mammals, breeding birds, amphibians, or reptiles in response to the prescribed fire.

Introduction

Oak (*Quercus* spp.) woodlands are the most extensive vegetation type in California, covering approximately three million ha (Rossi 1979; Griffin and Muick 1984). In varying compositions, 10 native shrub and nine native tree species comprise numerous woodland habitat types. More vertebrate wildlife species use oak woodlands than any other vegetation type in California (Ohmann and Mayer 1987; Airola 1988).

California oak woodlands are fire-adapted, having evolved with fire during the past one million years. The dominant use of oak woodlands is rangeland for livestock production. Prescribed fire is used in oak woodlands as a livestock forage and fuel management tool (Griffin and Muick 1984). The California Fire Plan (California Department of Forestry and Fire Protection 1996) suggests increased use of prescribed fire in California's oak woodlands to reduce the severity of wildfires by limiting fuel accumulation and to manage livestock forage.

In the central coast region of California (roughly the area between Santa Barbara and San Francisco extending to the coast mountain ranges approximately 80 km inland), the California Department of Forestry and Fire Protection (CDF) conducts prescribed burns on 1,000–4,000 ha annually (Ben Parker, CDF, San Luis Obispo Ranger Unit, San Luis Obispo, CA, pers. comm.). The CDF presumes that prescribed fire benefits wildlife of oak woodland habitats, but no published research supports this assumption. Most research on effects of prescribed fire to wildlife and their habitats in California historically has been conducted in chaparral ecosystems (Lawrence 1966; Lillywhite 1974; Longhurst 1978; Quinn 1979, 1983; Wirtz 1979, 1982). Except for a manuscript from this project (Vreeland and Tietje 1998), we are aware of no published research on effects of prescribed fire to California oak woodland habitats or associated wildlife species. Our objective was to quantify vegetation change and numerical response of small, nongame vertebrates to a prescribed fire conducted in mixed oak woodlands of the central coast region of California.

Study Area

We conducted this study at Camp Roberts (CR), California, a facility of the California Army National Guard located approximately 30 km from the Pacific Ocean in northern San Luis Obispo and southern Monterey counties. Topography at CR varies from flat to gently rolling hills and steep (>45° slope) hills. The climate of the study area is Mediterranean, characterized by cool, wet winters and hot, dry summers. Annual precipitation averages 38 cm (66-year range=10.8–74.1 cm), falling almost exclusively as rain between November and March. Mean annual temperature averages 15.3°C. Summer high temperatures frequently approach 50°C; winter lows rarely fall below –6°C. Sheep grazing and military training occur in our study area, but are limited in extent, duration, and intensity. Fire has been excluded from the study area for >15 years (Brian Duke, CR Environmental Office, Camp Roberts, CA). Our study sites were on slopes <20°, on north- to east-facing aspects, and 300–500 m elevation.

Camp Roberts covers 17,000 ha, with 41% classified as oak woodland (Camp Roberts EMAP 1989). Three oak habitat types occur at CR: valley oak (*Q. lobata*), coast live-oak (*Q. agrifolia*), and blue oak (*Q. douglasii*), with considerable overlap between coast live-oak and blue oak types.

We used blue oak and mixed blue oak–coast live oak stands in the San Luis Obispo County portion of CR (Tietje et al. 1997; Tietje and Vreeland 1997*a*). Blue oak was the dominant canopy species with a variable contribution (0–45%) of coast live-oak. Where present, understory vegetation was composed primarily of toyon (*Heteromeles arbutifolia*), redberry (*Rhamnus crocea*), poison oak (*Toxicodendron diversilobum*), and bigberry manzanita (*Arctoctsphylos glauca*). Small (≤ 0.25 ha) patches of chaparral (*Ceonothus* spp. and *Adenostoma fasciculatum*) occurred on three study plots. Common forbs included hummingbird sage (*Salvia spathacea*), deerweed (*Lotus scoparius*), filaree (*Erodium* spp.), and fiddleneck (*Amsinckia* spp.). Wild oats (*Avena* spp.),

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[1] Pennsylvania Cooperative Fish and Wildlife Research Unit, University Park, PA 16801

[2] Department of Environmental Science, Policy, and Management, University of California, Berkeley, CA 94720

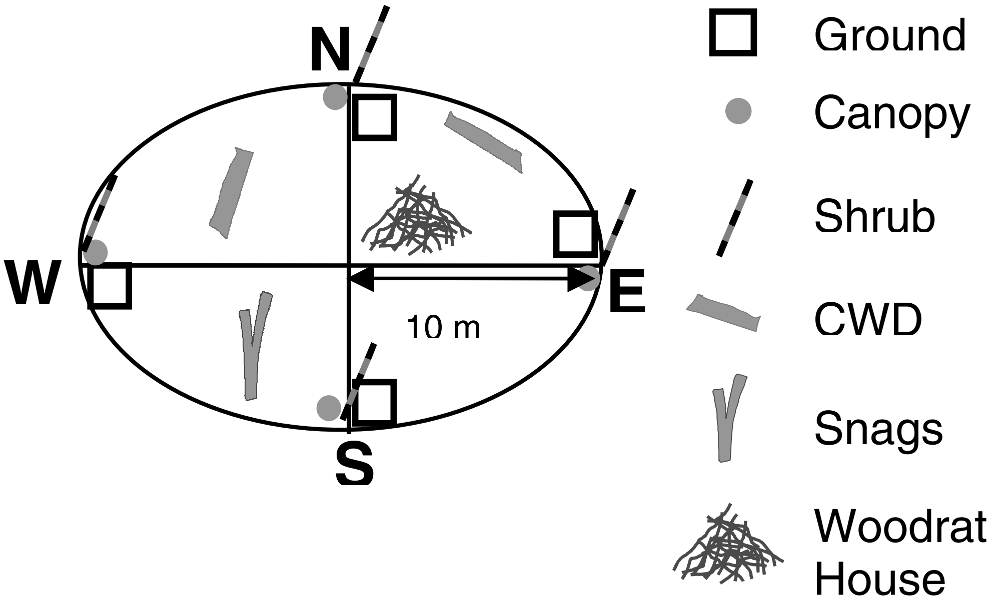

Figure 1.—Schematic layout of 10-m, circular plots used to sample vegetation characteristics in blue oak–coast live oak woodlands in San Luis Obispo County, California, 1997–1998. "Ground" represents ocular estimation of ground cover in 1 m^2 quadrats. "Canopy" represents measurement of canopy cover by concave spherical densiometer (Lemmon 1956). "Shrub" represents measurement of understory obstruction or shrub cover with a vegetation sampling pole (Griffith and Youtie 1988). "CWD," "Snags," and "Woodrat House" represent counting these elements within 10 m of plot centers. Not drawn to scale.

bromes (*Bromus* spp.), and fescues (*Festuca* spp.) dominated grassy openings of the woodland floor.

Blue oak sites were characterized by 40–60% canopy cover, <10% shrub cover, and >50% ground cover of exotic annual grasses. Mixed blue oak–coast live-oak sites were characterized by 60–90% canopy cover, 10–30% shrub cover, and a well-developed, thick leaf litter layer with abundant herbs, forbs, and <30% grass cover. Tietje et al. (1997) and Tietje and Vreeland (1997*a*) give detailed descriptions of vegetative and habitat characteristics of our study sites.

Methods

Prescribed Fire

We selected two areas to receive a prescribed fire treatment. One area (hereafter "Burn 1") was 80 ha, the other area (hereafter "Burn 2") was 120 ha. The California Department of Forestry and Fire Protection burned these areas on consecutive days in October 1997. The perimeter was burned first, then the interiors were ignited using drip torches, and accelerants delivered from helicopters. Surrounding unburned areas hereafter are referred to as "Ctrl" or "unburned areas."

Vegetation Sampling

We sampled vegetation in 11 1-ha plots throughout the burned areas within two months before and within 2–3 months after the prescribed fire. We randomly located five 10-m-radius sampling stations on each plot (Figure 1). At 10 m in the cardinal directions from the central point of each station, we measured canopy cover with a concave spherical densiometer (Lemmon 1956), understory obstruction (an index to shrub cover) with a vegetation pole (Griffith and Youtie 1988) graduated in five 0.5-m sections, and ocularly estimated ground cover within 1-m^2 frames. Within 10 m of the central point of each station, we counted the number of pieces of coarse woody debris (CWD, ^{3}1 m long and maximum diameter ^{3}10 cm), number of snags (^{3}1.37 m tall and ^{3}10 cm diameter at breast height), and number of dwellings (hereafter "houses") constructed by dusky-footed woodrats (*Neotoma fuscipes*).

Small Mammal Sampling

We live-trapped small mammals on 22 1-ha grids (8 x 8 grids, 15-m spacing) during May, 1997–1999. Plots were evenly divided between burned and unburned areas; six plots were in Burn 1 and five plots were in Burn 2. Plots were ^{3}42 m apart. Spatial separation between plots was adequate because <4 animals moved among plots during the course of the study (W.D. Tietje, unpublished data). We trapped for five consecutive nights during 1997 and three consecutive nights during 1998 and 1999. We baited traps with horse feed ("COB": corn, oats, and barley laced with molasses), tagged animals with individually numbered ear tags, and released them at site of capture. We handled animals in accordance with University of California, Berkeley Animal Use Protocol #R166-0199.

Breeding Bird Sampling

We conducted point-counts of breeding birds during March–April, 1997–1999. We used 86 50-m-radius point count stations located ^{3}150 m apart. Stations were evenly divided between burned (13 stations were in Burn 1, 30 stations were in Burn 2) and unburned areas. We visited each station six times in each season. We rotated station visitations among start time, observer, and treatment. Two to four observers conducted counts in each year. We conducted counts for 10 minutes at each station between official sunrise and 1100 hours using standard breeding-bird survey protocols (Bibby et al. 1992).

Amphibian and Reptile Sampling

We counted amphibians and reptiles under 136 61-cm x 61-cm x 1.27-cm plywood coverboards (DeGraaf and Yamasaki 1992; Grant et al. 1992; Tietje and Vreeland 1997*b*) on each of nine 5.8-ha plots (8 x 17 grid, 30- x 15-m spacing). Five plots were in the unburned area, two plots each were in Burn 1 and Burn 2. We checked each coverboard once every 7–10 days during late January–early May in 1995–1999. We first deployed coverboards in 1994 to allow them to weather for 6–9 months, dissipating chemicals used in their manufacture that might affect their use by amphibians and reptiles (Grant et al. 1992). We had difficulty identifying slender salamanders (*Batrachoseps* spp.) (N. Scott, U.S. Geological Survey, Biological Resources Division, San Simeon, CA, pers. comm.). Pacific slender salamanders (*B. pacificus*) and black-bellied salamanders (*B. nigriventris*) potentially occur at CR, but cannot be distinguished in the field. Therefore we grouped them into one species category, hereafter "slender salamanders."

Analyses

We qualitatively described fire behavior and counted the number of grid intersections on small mammal trapping grids where fire burned to within 1 m to quantify areal coverage of the fire. In Vreeland and Tietje (1998), we conducted paired-sample *t*-tests to assess vegetation changes and considered differences significant at $\alpha=0.05$. We repeat an abbreviated version of those vegetation results here, but direct the reader to Vreeland and Tietje (1998) for details on vegetation results. Because we were unable to randomly select burned areas and because treatment replication was low, our animal data cannot meet some assumptions of analyses of variance (ANOVA). Accordingly, rather than conduct inappropriate, low-power statistical tests, we used means and 95% confidence intervals (CI) to assess numerical response of vertebrates to the prescribed fire. We selected one species to represent each of the four taxa we monitored. We selected a representative species based on two characteristics: large relative abundance and potential for demonstrating change as a result of prescribed fire. True replication in our study is at the burn unit level. Relative abundance was calculated for each sampling unit (plot or point count station), then averaged for a treatment area (Burn 1, Burn 2, Ctrl). Because we have pre- and post-fire data on burned and unburned areas, pre-fire differences in relative abundance between burned and unburned areas are unimportant. The meaningful test of an effect of the fire is a comparison between relative abundance of a species on burned areas compared to unburned areas before and after the fire. This is analogous to the interaction term in a 2-way ANOVA. Differences between treatment means were considered significant if 95% of the CI did not overlap.

Results

Prescribed Fire

The California Department of Forestry and Fire Protection rated the prescribed fire as light- to moderate-intensity (3–4 on a 10-point scale). Flame height generally was <1 m except in a few areas of decadent grasses, chaparral, or dense accumulations of CWD. Few mature trees died. The fire was patchily distributed and carried better through blue oak stands with abundant grass cover than through mixed oak stands with thick leaf litter and dense canopy cover. Mineral soil was mostly unexposed except in small areas with high fuel load (dense CWD, chaparral). Area burned on 1-ha plots averaged 46% and ranged from 30 to 66%.

Vegetation

Understory obstruction and grass cover were reduced by 7% and 70%, respectively. Much of the reduction in understory obstruction was the result of reduction in grass cover that covered the lower 1 m of the vegetation pole. Excluding this grass cover probably would result in no statistical reduction in understory obstruction. Grass cover returned to pre-fire cover by one growing season after the fire. Canopy cover, number of snags, and leaf litter depth (an index to litter volume), did not change. Approximately six % of the canopy was singed. Number of woodrat houses was reduced by 30%. Number of pieces of CWD was reduced by 35%. We recorded five new pieces of CWD in the 55 10-m vegetation plots. These pieces of CWD were created when roots of mature oaks were burned during the fire, causing the bole to fall to the ground. Vreeland and Tietje (1998) report detailed vegetation responses to the prescribed fire.

Small Mammals

We captured nine species of small mammals during three years of live-trapping (Table 1). Woodrats, piñon

Table 1.—Relative abundance of small mammals, breeding birds with at least 100 detections, amphibians, and reptiles monitored in response to prescribed fire in blue oak–coast live oak woodlands in San Luis Obispo County, California, 1995–1999

Common name (scientific name)	Relative abundance[a]
Small Mammals	
Dusky-footed woodrat (*Neotoma fuscipes*)	26.49
Piñon mouse (*Peromyscus truei*)	6.61
Brush mouse (*Peromyscus boylii*)	4.40
California mouse (*Peromyscus californicus*)	3.06
California pocket mouse (*Perognathus californicus*)	1.34
California vole (*Microtus californicus*)	0.36
Merriam's chipmunk (*Tamias merriami*)	0.36
Deer mouse (*Peromyscus maniculatus*)	0.11
Heermann's kangaroo rat (*Dipodomys heermanni*)	0.01
Breeding Birds[b]	
Oak titmouse (*Parus inornatus*)	3,001
Dark-eyed junco (*Junco hyemalis*)	2,564
Lesser goldfinch (*Carduelis psaltria*)	1,842
Bushtit (*Psaltriparus minimus*)	1,529
Western scrub-jay (*Aphelocoma coerulescens*)	1,344
House finch (*Carpodacus mexicanus*)	1,336
Bewick's wren (*Thryomanes bewickii*)	1,221
Violet-green swallow (*Tachycineta thalassina*)	805
Spotted towhee (*Pipilo erythrophthalmus*)	643
Anna's hummingbird (*Calypte anna*)	517
California quail (*Callipepla californica*)	505
White-breasted nuthatch (*Sitta carolinensis*)	485
Hutton's vireo (*Vireo huttoni*)	480
Yellow-rumped warbler (*Dendroica coronota*)	480
Golden-crowned sparrow (*Zonotrichia atricapilla*)	459
Ash-throated flycatcher (*Myiarchus cinerascens*)	453
California towhee (*Pipilo crissalis*)	426
Western bluebird (*Sialia mexicana*)	405
Nuttall's woodpecker (*Picoides nutallii*)	394
Mourning dove (*Zenaida macroura*)	350
Ruby-crowned kinglet (*Regulus calendula*)	315
Blue-gray gnatcatcher (*Polioptila caerulea*)	285
Acorn woodpecker (*Melanerpes formicivorus*)	246
Orange-crowned warbler (*Vermivora celata*)	245
Wrentit (*Chamaea fasciata*)	204
Lawrence's goldfinch (*Carduelis lawrencei*)	177
Hermit thrush (*Catharus guttatus*)	141
Lark sparrow (*Chondestes grammacus*)	134
Northern flicker (*Colaptes auratus*)	121
Brown-headed cowbird (*Molothrus ater*)	111
Amphibians and Reptiles	
Western skink (*Eumeces skiltonianus*)	3.96
Western fence lizard (*Sceloporus occidentalis*)	2.19
Slender salamander (*Batrachoseps* sp.)	1.74
California legless lizard (*Anniella pulchra*)	0.47

continued

Table 1.—continued

Common name (scientific name)	Relative abundance[a]
Southern alligator lizard (*Gerrhonotus multicarinatus*)	0.35
Gopher snake (*Pituophis melanoleucus*)	0.22
California whipsnake (*Masticophis lateralis*)	0.22
Ring-necked snake (*Diadophis punctatus*)	0.14
Night snake (*Hypsiglena torquata*)	0.07
Common king snake (*Lampropeltis getulus*)	0.06
Chorus frog (*Hyla regilla*)	0.06
Side-blotched lizard (*Uta stansburiana*)	0.03
Western toad (*Bufo boreas*)	0.02
Common garter snake (*Thamnophis sirtalis*)	0.01
Monterey salamander (*Ensatina eschscholtzii*)	0.01

[a] Small mammals: captures/100 trap-nights; breeding birds: total observations during 3 years (86 total point count stations); amphibians and reptiles: observations/100 coverboard checks.

[b] Fifty-five species of birds were observed fewer than 100 times during 3 years and are not shown.

mice (*Peromyscus truei*), brush mice (*P. boylii*), and California mice (*P. californicus*) were the four most abundant species. Woodrats were the most frequently captured species (26.5 captures/100 trap-nights during 1997–1999). Relative abundance (captures/100 trap-nights) of woodrats on burned areas compared to the unburned areas did not change after the prescribed fire (Figure 2, Table 2). Relative abundance of woodrats was approximately 60% lower in 1998 and 1999 than in 1997, but this reduction was proportional among burned and unburned areas. Confidence intervals consistently overlapped treatment means and other CI within and among years.

Breeding Birds

We observed over 85 species of birds during the breeding season on all 86 point count stations (Table 1). Dark-eyed juncos (*Junco hyemalis*) were one of the two most frequently observed species (1.7 observations/point/visit during 1997–1999, respectively). Relative abundance (mean number of observations/point) of juncos on burned areas compared to unburned areas did not change after the prescribed fire (Figure 3, Table 2). We observed an approximately 80 % increase in juncos two years post-fire, but this increase was proportional among burned and unburned areas. Juncos consistently were more abundant on Burn 1 than Ctrl, but the difference remained consistent before and after the fire. Other confidence intervals were narrow, but consistently overlapped treatment means and other CI within and among years.

Amphibians and Reptiles

We observed four amphibian (two salamanders, one frog, one toad) and 11 reptile (five lizards, six snakes) species under plywood coverboards (Table 1). Among amphibians, only slender salamanders were observed in appreciable numbers (1.74 observations/100 coverboard visits during 1995–1999). Relative abundance (number of observations/100 board-visits) of slender salamanders was similar on burned areas compared to unburned areas before and after the prescribed fire (Table 2). Except during 1999, slender salamanders were more abundant on Burn 1 than Burn 2 or Ctrl, but this pattern of relative abundance remained consistent among years before and after the prescribed fire. Confidence intervals were broad, frequently included zero and other treatment means, and overlapped other CI within and among years.

Western skinks (*Eumeces skiltonianus*) were the most frequently observed reptile species (3.96 observations/100 coverboard visits during 1995–1999). Relative abundance (number of observations/100 board-visits) of western skinks varied among years and among treatment areas within and among years, but was similar on burned areas compared to unburned areas before and after the prescribed fire (Table 2). Skink observation rate appeared to be increasing on Ctrl, decreasing on Burn 2, and irregular on Burn 1, but these trends began 2–3 years before the fire, and were not caused by nor interrupted by the fire. Confidence intervals were broad, frequently included zero and other treatment means, and overlapped other CI within and among years.

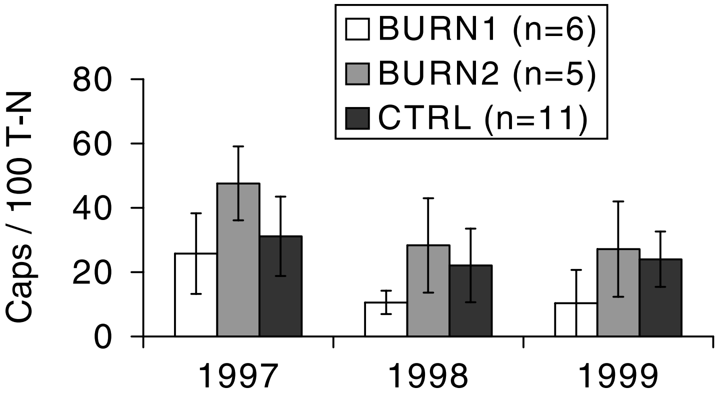

Figure 2.—Dusky-footed woodrat abundance (captures per 100 trap-nights) in May 1997–1999 before and after an October, 1997 prescribed fire in blue oak–coast live oak woodlands in San Luis Obispo County, California. Error bars are 95% confidence intervals.

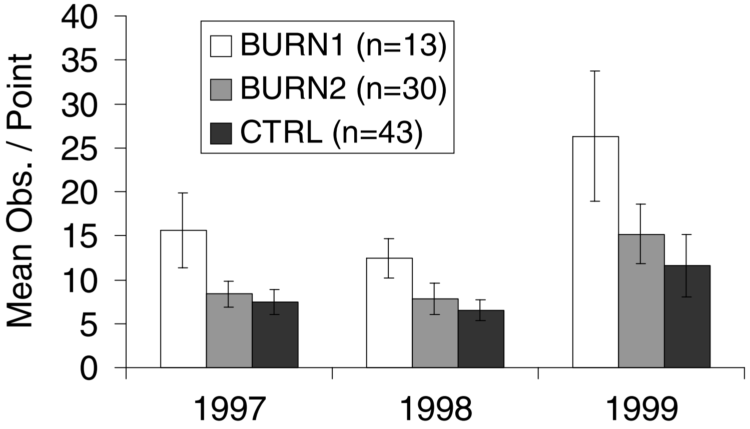

Figure 3.—Dark-eyed junco abundance (mean number of observations per point) in March and April, 1997–1999 before and after an October, 1997 prescribed fire in blue oak–coast live oak woodlands in San Luis Obispo County, California. Error bars are 95% confidence intervals.

Discussion

Conclusions

We detected no change in relative abundance of small mammals, breeding birds, amphibians, or reptiles after a light- to moderate-intensity prescribed fire in California blue oak–coast live oak woodlands. Intensity of the fire was limited by cool temperatures, moderately high relative humidity, low wind speed, and resulted in generally marginal or short-term changes to the vegetation characteristics we measured. Prescribed fire in oak woodlands may reduce resource competition from exotic annual grasses, stimulating shrub and tree health and vigor and, ultimately, mast production and overall habitat rejuvenation. Small vertebrates may respond to this habitat rejuvenation.

Coarse woody debris serves as hiding, breeding, and foraging habitat for small vertebrates. Reduction in CWD was substantial after the prescribed fire, but CWD may not be limiting for animals that use it. We counted 237 pieces of CWD on 1.73 ha of vegetation sampling plots (137 pieces/ha). Size criteria for CWD was appropriate to gauge use by most vertebrates but is a conservative estimate of CWD abundance, because small vertebrates will use smaller pieces of CWD. We observed slender salamanders using pieces of woody debris (e.g., dead coast live-oak bark) as small 20 cm x 20 cm x 3 cm, considerably smaller than the minimum size of CWD for this study.

Our casual observations detected only one instance (a woodrat) of direct mortality caused by the fire. Direct mortality likely was low, because most animals we monitored have subterranean access or are capable of other strategies that enable them to escape fire. Russell et al. (1999) reviewed over 12 published papers on effects of fire to herpetiles and concluded that direct mortality of these animals in fires is low.

No other published literature reports on the effects of prescribed fire to vegetation structure or vertebrate communities in California oak woodlands. Many researchers have compared burned and unburned areas after fire, but lack of pre-fire observations limits conclusions that can be drawn about effects of fire to animals. Some studies have combined other silvicultural treatments (e.g., thinning, forest harvesting) with burning. Distinguishing effects of burning from other treatments can be difficult, especially if treatments were not conducted under mutually exclusive study designs. Much study of prescribed fire and its effects on habitat

Table 2.—Relative abundance (mean, 95% CI) of dusky-footed woodrats, dark-eyed juncos, slender salamanders, and western skinks on burned and unburned areas before and after prescribed fire in blue oak–coast live oak woodlands in San Luis Obispo County, California, 1995–1999. The prescribed fire was conducted in October, 1997. Pre-fire years are 1995–1997 for salamanders and skinks, and 1997 for woodrats and juncos. Woodrats and juncos were not monitored in 1995 and 1996. See text for sample sizes

Year	Treatment	Dusky-footed woodrat	Dark-eyed junco	Slender salamander	Western skink
1995	Burn 1			8.6 (0–89.0)	2.6 (00.–13.3)
	Burn 2			0.8 (0–07.7)	4.0 (00.–08.3)
	Ctrl			0.6 (0–01.4)	3.3 (00.–06.7)
1996	Burn 1			2.0 (0–15.1)	2.0 (00.–12.3)
	Burn 2			1.0 (0–11.3)	3.4 (00.–10.4)
	Ctrl			0.3 (0–00.6)	3.5 (00.–07.0)
1997	Burn 1	25.8 (13.2–38.3)	15.6 (11.3–19.9)	2.9 (0–19.2)	1.7 (00.–07.7)
	Burn 2	47.6 (36.1–59.1)	08.4 (06.9–09.8)	2.4 (0–21.6)	2.7 (00.–09.7)
	Ctrl	31.2 (18.8–43.5)	07.4 (06.0–08.9)	1.0 (0–02.2)	4.8 (0.2–09.3)
1998	Burn 1	10.6 (07.0–14.2)	12.4 (10.1–14.6)	6.6 (0–52.4)	3.3 (00.–26.1)
	Burn 2	28.3 (13.6–43.0)	07.8 (06.0–09.6)	3.2 (0–32.1)	2.4 (00.–04.7)
	Ctrl	22.1 (10.6–33.6)	06.5 (05.4–07.6)	1.0 (0–02.2)	6.7 (0.3–13.0)
1999	Burn 1	10.3 (00.0–20.7)	26.3 (18.9–33.7)	1.8 (0–15.4)	2.8 (00.–18.9)
	Burn 2	27.2 (12.4–42.0)	15.9 (11.8–18.6)	2.8 (0–33.1)	2.4 (00.–15.6)
	Ctrl	24.0 (15.4–32.7)	11.6 (08.0–15.2)	0.4 (0–00.9)	6.7 (00.–12.7)

structure and wildlife have been conducted in other ecosystems (e.g., California chaparral, California dry pine, southeastern U.S. pine forests, prairie, northern temperate oak savannah, Appalachian hardwoods, and others). Species in different habitats and ecosystems likely respond differently to prescribed fire; therefore, comparisons of our results to other prescribed fire studies in other habitat types are hampered by these cross-habitat differences. Therefore, we restrict our comparisons with other prescribed fire studies to those that experimentally investigated effects of fire, rather than with studies that conducted post-fire observations on burned and unburned areas, on burned areas before and after fire without unburned controls, and wildfires that generally burn hotter and faster than prescribed fires.

Despite a 30% reduction in number of woodrat houses, we observed no change in relative abundance of woodrats because of the prescribed fire. Not all woodrat houses are occupied (Vreeland and Tietje 1999) and fire did not reach many houses in the densest habitats; therefore, few woodrats were forced to relocate to new areas or construct new houses after the fire. Tevis (1956) experimentally investigated effects of a site-preparation fire in logging debris (Pacific madrone [*Arbutus menziesii*] and tanoak [*Lithocarpus densiflorus*]) on numbers of small mammals in northern California Douglas-fir (*Pseudotseuga menziesii*) stands. Over 70 % of marked animals were not recaptured after the fire and probably died or emigrated from the burned area immediately after the fire. Most recaptured animals were caught at the periphery of the fire, suggesting they were not resident within the burn blocks before the fire or temporarily relocated after the fire. However, within three weeks after the fire, more new individuals were captured in the burned area than were captured before the fire. Tevis (1956) attributed the immigration by new individuals to the lack of resident mice created by the fire. We observed no such immigration to burned areas, probably because few or no residents were killed by the fire or emigrated from the burned areas, thereby leaving little unoccupied habitat.

Similarly, Tester (1965) trapped deer mice (*P. maniculatus* and *P. leucopus*) before and after a prescribed fire in burr oak (*Q. macrocarpa*)–northern pin oak (*Q. ellipsoidalis*) savanna in Minnesota and reported substantial immigration to burned areas within one month after the fire. Few animals captured before the fire were recaptured after the fire, but Tester (1965) could not speculate if absence of these animals suggested low survival during the fire, emigration, or high predation following the fire. Tester (1965) attributed increased use of burned areas to reduction in litter cover and depth, which were negatively correlated with presence of deer mice.

Masters et al. (1998) observed greater small mammal abundance, species richness, and diversity in Arkansas shortleaf pine (*Pinus echinata*) stands treated with prescribed fire than in unburned stands. Sullivan and Boateng (1996) observed dramatic decreases in deer mice and Oregon voles (*Microtus oregoni*) immediately following prescribed fire in coniferous forests of British Columbia, but animal abundance returned to pre-fire levels within four months following the fire. Red-backed voles (*Clethrionomys gapperi*) also increased substantially immediately following fire. Kaufman et al. (1988) observed a doubling of deer mouse abundance on burned areas 3–5 weeks following prescribed fire in Kansas tall-grass prairie. Ford et al. (1999) observed a decrease in abundance of pine voles (*Microtus pinetorum*) following prescribed burning in pitch pine (*Pinus rigida*) and oak stands in North Carolina, but abundance of nine other species did not change following fire. Lawrence (1966) trapped dusky-footed woodrats and observed immediate, but marginal short-term reductions in abundance following prescribed fire in chaparral and grasslands in California. Pattern of abundance between burned and unburned areas in both vegetation types remained similar for nearly three years following fire. Lawrence (1966) also observed decreased mass of piñon mice on burned areas following fire, but seasonal pattern of mass was similar between burned and unburned areas. Lawrence (1966) attributed decreased mass to 90% reduction in food abundance. Masters et al. (1998) and Kaufman et al. (1988) attributed increases in abundance to more favorable habitat or foraging conditions in burned areas.

Effects of prescribed fires conducted during fall may be less extensive or severe for breeding bird communities than for wintering bird communities in California oak woodlands, because breeding activity and behavior may commence during or after habitat rejuvenation initiates. Wilson et al. (1995) observed varied responses by breeding birds to prescribed fire in shortleaf pine–oak forests in Arkansas. Overall, bird densities and density of ground- and shrub-foraging and shrub-nesting species were greater in burned than in unburned stands following fire. Similarly, Salveter et al. (1996) observed no changes in abundance of neotropical migrant or resident songbird species following experimental prescribed fire in shortleaf pine and oak forests in Arkansas. Lawrence (1966) observed immediate reductions in abundance of chaparral birds following prescribed fire in California chaparral. Lower bird abundance persisted in burned areas compared to unburned areas through three years post-fire. Conversely, Lawrence (1966) observed increases in grassland birds and no significant change in abundance of oak woodland birds during the same fire in chaparral. Horton and Mannan (1988) observed a reduction in abundance of northern flickers (*Colaptes auratus*) and violet-green swallows (*Tachycineta thalassina*), an increase in mountain chickadees (*Parus gambeli*), and no change in 13 other species following a low-intensity prescribed fire in Arizona ponderosa pine (*Pinus ponderosa*) forests. We also monitored violet-green swallows and flickers in our oak-woodland study site, but observed no changes in their relative abundance, although detection rates were low. Salveter et al. (1996) suggested that some guilds (e.g., ground foragers, cavity nesters) may collectively demonstrate increases or decreases in abundance following prescribed fire. Reynolds and Krausman (1998) observed no reduction in relative abundance of breeding birds or wintering birds following experimental winter prescribed burning in Texas mesquite (*Prosopis glandulosa*) grasslands, and argued that species respond differently, potentially confounding guild analyses. Petersen and Best (1987) observed no immediate reduction in bird density following prescribed fire in sagebrush (*Artemisia* spp.) rangelands in Idaho, but documented increased species richness and density two years post-fire. Winter and Best (1985) observed a change in placement position of sage sparrow (*Amphispiza belli*) nests following prescribed burning in sagebrush (*Artimesia* spp.) rangelands in Idaho, suggesting functional responses to prescribed fire may be as important as changes in bird abundance.

Slender salamanders require moist microhabitats to maintain suitable respiratory conditions, and therefore are closely associated with habitat components (e.g., dense canopy cover, presence of CWD) conducive to maintaining these conditions. Western skinks are habitat generalists and ubiquitous, although they likely require CWD as refugia (Tietje et al. 1997; Tietje and Vreeland 1997*a*). Because CWD was abundant on our study plots, CWD probably is not limiting populations of slender salamanders, western skinks, or other herpetiles. Therefore, even large reductions in CWD may not result in detectable changes in relative abundance of small vertebrates.

We were unable to find experimental studies on effects of prescribed fire (without other silvicultural treatments) to amphibians and reptiles. Anecdotal and observational studies suggest that amphibians and reptiles may respond negatively (Enge and Marion 1986, McLeod and Gates 1998) or not at all (Enge and Marion 1986, Ford et al. 1999) to prescribed fire. Other observational studies suggest reptiles may exhibit different functional or behavioral activity in burned areas (Kahn 1960, Lillywhite and North 1974, Lillywhite et al. 1977). Observation rates of amphibians and reptiles typically are low in most studies, contributing to low measurement precision in response to treatments.

Because we did not mark amphibians and reptiles, we relied on observation rates to quantify their relative abundance. If herpetile numbers were reduced by the fire, these reductions could be masked by increased movement and subsequent increased observation rates, or increased use of coverboards with fire-induced loss of natural cover objects. Increased movement rates could result from changes in prey (e.g., small mammals, invertebrates) abundance, foraging behavior, increased ease of mobility through ground vegetation reduced by fire, or genuine increases in animal abundance.

Caveats

Replication in our study was low because plots and point-count stations are pseudoreplicates of the burned (n=2) and unburned (n=1) areas. Therefore, true replication in our study was the burn blocks. Replicating prescribed fire studies at this scale is difficult for logistical reasons and budget constraints, but researchers should strive to replicate at the landscape level.

Sampling units were few for amphibian and reptile data (four burned plots, five unburned plots), contributing to low statistical power (broad CI) for these data. Furthermore, we did not mark individual birds, amphibians, or reptiles; therefore, our measures of relative abundance lack precision and may overestimate relative abundance. Although we individually marked small mammals, budget constraints restricted us to trapping only three nights during the two post-fire trapping sessions, which forced us to use capture rates rather than number of individuals as the metric for small mammal relative abundance. Data collected on number of individuals rather than capture rates has lower explanatory power.

Research on breeding birds suggests that bird density and relative abundance are poor measures of habitat quality and indicators for responses to treatments (Van Horne 1983, Vickery et al. 1992*a,b*). Engstrom et al. (1996) suggest that estimates of productivity and site fidelity would be better variables than density or relative abundance for demonstrating potential effects of prescribed fire to avian communities because relative abundance or density can be an inappropriate index to habitat quality.

Despite these shortcomings to our study, we believe our research represents important, initial information on effects of prescribed fire to blue oak–coast live-oak woodland habitat and associated small, nongame vertebrates. In addition, consistent responses to the prescribed fire by species in the four taxa we monitored suggest that our conclusions are appropriate.

We conducted the prescribed fire in October 1997, but assessed responses of small vertebrates 3–7 months after the fire. Monitoring animals immediately after the fire may have resulted in different effects to our study taxa. Our observations on the response to prescribed fire by small mammals during fall (October and November) and by wintering birds (January and February) followed similar patterns to the monitoring efforts we report here, however (Vreeland and Tietje 1998; W.D. Tietje, unpubl. data).

Research Implications

Studies of effects of prescribed fire must maintain an experimental design: before and after sampling on burned and unburned areas is required to document changes in animal numbers on burned areas compared to unburned controls (Russell et al. 1999). Replicating at landscape-level scales (>100 ha) is critical to assessing effects of prescribed fire, which necessarily occurs over large areas. Effects of prescribed fire, particularly for more intense fires, may last beyond two years post fire; long-term studies should evaluate effects of prescribed fire >5–20 years post-fire. Response of vegetation and animals may differ among fire intensity and among seasons in which prescribed fires are conducted (Engstrom et al. 1996, King et al. 1998). Further study of effect of season and intensity of fire is warranted. Researchers should mark individuals, preferably with individual marks, to precisely measure changes in numbers of individuals after prescribed fires. In addition to numerical responses, animals may demonstrate functional and behavioral responses to prescribed fire (Kahn 1960, Lillywhite and North 1974, Lillywhite et al. 1977). Researchers should assess functional response of animals and changes in behavior (foraging and nesting behavior, movement patterns, home range size) as well as numerical response (Winter and Best 1985, Engstrom et al. 1996). Estimates of reproductive potential (nest success [Engstrom et al. 1996], small mammal testes size), or individual status (health index [Lawrence 1966]) before and after prescribed fire would greatly enhance knowledge of effects of prescribed fire to small vertebrates (Engstrom et al. 1996).

Acknowledgements

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Fire Management and Research for Biodiversity in the Green Swamp

Margit A. Bucher[1] and Maura E. High[1]

Abstract.—The Nature Conservancy has conducted prescribed burns for two decades in North Carolina's Green Swamp to enhance and maintain species biodiversity in the longleaf pine (*Pinus palustris*) savannas and ecotones. The surrounding pocosins, however, have been left alone. Increasingly, dense shrub growth in the pocosin suppresses herbaceous species and increases the risk of serious wildfires. Two experimental burns have shown that it is possible to safely conduct prescribed burns in pocosins. Continuing studies will monitor the vegetation response to fire and map the vegetation and fuels to predict fire behavior, leading to a better fire management plan for the Green Swamp based on its historic fire regime.

In managing more than 1,500 natural areas (6.8 million acres) throughout the United States, The Nature Conservancy (TNC) examines the processes that have shaped and are shaping the ecosystems targeted for conservation within its program. The Nature Conservancy uses prescribed fires where this tool will contribute to the perpetuation of species and ecosystems. The Nature Conservancy has increased its ecological burning for ecological reasons to just over 70,000 acres, or 450 burns per year, in 44 states over the last decade. Fire exclusion and altered fire regimes have been identified as threats in 45% of sites where TNC actively is working and over 600,000 acres presently are targeted for burning. The North Carolina TNC Chapter has 14 preserves with 25,000 acres under fire management, ranging from longleaf and pond pine (*Pinus serotina*) ecosystems to Carolina bays, Piedmont prairie remnants, and a lone mountain seepage bog. In the 1980's, TNC began its fire management by burning longleaf pine savannas during the winter months. This is the traditional prescribed burning season in the region because during this time fire behavior is most predictable and controllable. As the chapter's staff gained experience with prescribed burning, a shift to conducting burns designed and timed to mimic fire regimes under which the natural communities evolved occured, also while protecting and promoting associated indigenous rare species. In North Carolina, this meant that the TNC's statewide prescribed burns are now conducted year-round, but with emphasis on the growing season to mimic wildfires as a key ecological process.

Role of Fire in the Ecology of the Green Swamp

The Green Swamp is a 17,000 acre preserve in the southeastern Coastal Plain of North Carolina. The majority of the preserve (about 85%) supports pocosin and pond pine woodlands on organic soil (Figure 1). Only a small portion of the preserve has mineral soils that support longleaf pine communities or loblolly pine (*Pinus* taeda) plantations (currently being restored back to longleaf pine communites). Fire and hydrology have shaped the longleaf pine/pocosin (shrub bog) ecosystems in the coastal Carolinas, including the Green Swamp, and creating assemblages of species (communities) with specific components and characteristics (Christensen 1981). For example, the Green Swamp is renowned for the high diversity of herbaceous species in savannas where more than 30 species per square meter can occur (Walker and Peet 1983), including endemics such as Venus flytrap *(Dionea muscipula)* and rough-leaf loosestrife *(Lysimachia asperulifolia)*. Frequent fires (1–10 year return intervals) are critical to maintaining the open character of the savannas (Christensen 1981). These open conditions also are an important environmental factor for the health and persistence of associated rare species such as the red-cockaded woodpecker *(Picoides borealis)* and Venus flytrap moth *(Hemipachnobia subporphyrea) as noted by* Hall and Schweitzer (1993).

Fire naturally has occurred in the pocosins, occasionally burning into the peaty soils during drought periods (Hungerford et al 1998). Pocosin fires open up shrub areas allowing for a mixed-age mosaic of vegetation with a variety of herbaceous species, including the federally threatened rough-leaf loosestrife to become established. The historic fire regime under which pocosins developed is estimated between 5 to 30 years (Wells 1946; Frost 1995). Fire also stimulates root sprouting and reproduction and increases nutrient levels in the nutrient-poor soils of the pocosin (Schafale and Weakley 1990). The Green Swamp has a range of high and low pocosin communities, varying with respect to time since the last fire, the depth of their peat soils, the saturation and nutrient content of the soil, and the species they support. Low and high pocosins harbor such rare species as Cotton grass (*Eriophorum virginicum*), laurel (*Kalmia cuneata*), rough-leaved loosestrife, arrow arum (*Peltandra sagittifolia*), beak rush (*Rhynchospora alba*) and a sphagnum (*Sphagnum fitzgeraldii*) (Schafale and Weakley 1990).

In the absence of fire, all the Green Swamp communities lose species diversity and build up accumulations of fuels that can lead to a risk of more intense and more damaging wildfires. Moreover, with fire suppression, hardwood shrubs encroach on the formerly isolated savannas, shading out other species, with a similar loss of species that favor open conditions in the pocosins.

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[1] The Nature Conservancy, Durham, NC 27707

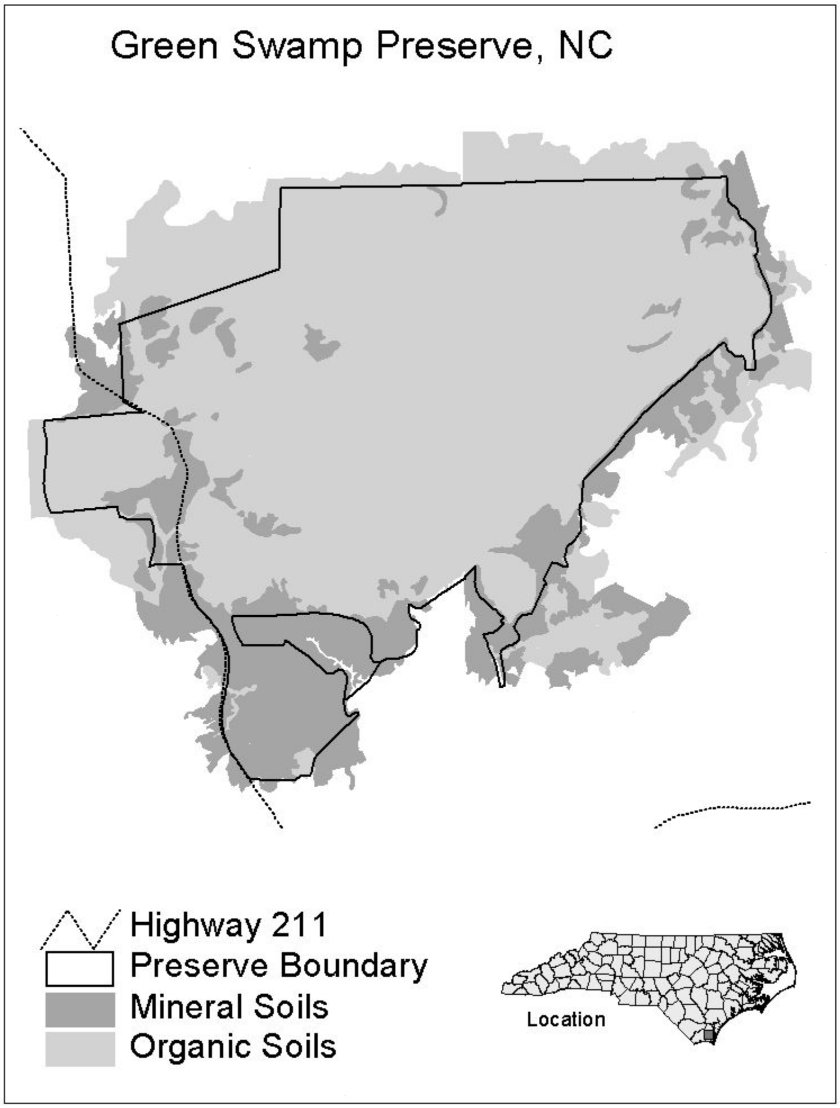

Figure 1.—Soil type and preserve boundary of the Green Swamp, North Carolina

The majority of the Green Swamp last burned during a 1932 wildfire; small portions of the pocosins burned in 1955, 1982, and 1990 as wildfires. The fire frequency on most of the preserve has been reduced through active fire suppression and lack of prescribed burning on a landscape level for the past 50 years. The heavy fuel accumulation (15-25 tons/acre) in the pocosins increases the risk of severe fires that could burn deeply into the peat during drought conditions.

The Nature Conservancy has conducted frequent prescribed burns in the longleaf pine savannas and pine plantations close to the highway since the early 1980s. Prescribed burning of savannas in the Green Swamp has been based on timber management burns during the dormant season. By burning within 2 days of a rain event, when the organic soils are saturated, the pocosins served as a natural fire breaks rather than establishing destructive plow lines. Such natural fire breaks avoid soil disturbance, especially in the ecotone between savannas and pocosin, where many rare and endangered occur. Most of the isolated, interior savannas and the pocosin, however, have burned only accidentally within the past century (Kologiski 1977, McIver 1981).

Developing a Fire Management Plan for the Green Swamp Preserve

The Nature Conservancy has become increasingly aware that the limited burning conducted in the past two decades was not consistent with stated organizational ecosystem conservation goals. Therefore, TNC developed the following fire management goals for the Green Swamp in 1995: (1) to restore and maintain communities in a spatially and temporally heterogeneous pattern using prescribed burns as a

Figure 2.—Aerial view of two experimental pocosin burns in the Green Swamp. Only the surface fuels burned in the foreground. The burn unit in the back shows the peat burn unit one year after a burn.

management tool to mimic fire regimes under which these communities evolved while protecting and promoting the rare species associated with them; (2) to shift from the traditional dormant season to year-round prescribed burning, but conducting more burns in the growing season to mimic wildfires as a key natural process; (3) to plan and time burns to restore and maintain the mosaic and diversity of habitats in different successional stages to support the species indigenous to the Green Swamp; (4) to expand prescribed burning to the entire preserve, and include isolated savannas and pocosins, at least on a experimental basis; (5) to use research and monitoring as part of the proposed fire management plan to evaluate the effectiveness of the prescribed burning regime and redirect management as necessary to accomplish the management goals (Bucher and Gintoli 2001).

It became apparent to TNC staff that to effectively manage the longleaf pine/pondpine ecosystems from an ecological as well as a logistical perspective requires prescribed burning in the savannas as well as in pocosins. Fire is needed in pocosin habitat not only to maintain its inherent diversity but also to reduce the threat and risk of wildfires originating on the preserve and threatening adjacent plantations that could lead to potential impact of suppression activities requiring heavy equipment. However, conducting safe prescribed burns in pocosin requires more information on prescribed burn parameters than is generally available.

The Nature Conservancy learned that the USDA Forest Service Intermountain Fire Science Lab (IFSL) was conducting a nationwide study examining prescribed burning issues in wetlands. In 1995, TNC developed a joint venture with the IFSL to use the Green Swamp as one of the research burn sites. The Green Swamp study goals are (1) to develop guidelines for prescribed pocosin burns that would not ignite organic soils and guidelines that could predict when peat fires would extinguish; (2) to understand the ecological consequences of suppressing peat fires versus allowing them to burn; and (3) to provide suppression agencies with better tools to predict the probability of severe peat fires on wildfires. In the course of this joint venture, IFSL staff gathered pre-burn peat moisture and surface topography data, sampled surface fuels, and monitored soil consumption and heat transfer during the burn. The Nature Conservancy staff gathered pre- and post-burn vegetation data. In cooperation with the North Carolina Division of Forest Resources (NCDFR), TNC conducted two experimental prescribed burns in the Green Swamp pocosin, on 9 September 1999 and 18 September 1999, to field test the predictive models for peat ignition developed by the IFSL, (Figure 2) document ignition patterns and heat transfer in organic soils and monitor

Figure 3.—Prescribed burn that burned into the Green Swamp pocosin and extinguished itself without suppression action.

vegetative response to peat burns and a surface burns to improve pocosin vegetation models (Bucher 1998). These carefully planned and implemented burns demonstrated that pocosins can be burned safely, even in drought conditions when a peat fires are more likely to occur. The data from these burns are currently being analyzed with published outputs planned by fall 2001. The two experimental burns (each about 10 acres) have provided data to customize fuel models and have helped establish parameters for burn plans, but methodologies to safely conduct prescribed burns on a larger operational scale of at least a several hundred acres are not yet realized.

In attempting to return fire to some of the isolated savannas, TNC staff observed significant differences in pocosin fuels and fire behavior in different areas of the preserve. Occasionally burns would move deeper into pocosin than anticipated, as in a February 1998 fire, whose path is shown on Figure 3. Rain was approaching as the fire burned in the pocosin. The fire was allowed to extinguish itself in the evening, after a flyover determined it would remain entirely on TNC property. Afterwards, TNC staff determined that there was a need for better vegetation and fuels information. In summer 2000, TNC began a vegetation mapping project of the Green Swamp and its surroundings with Duke University and the North Carolina Plant Conservation program. The goals of the mapping project were to capture the diversity in pocosin from an ecological as well as a fuels management perspective and develop classification methods that could easily be exported to other sites. We employed a remote sensing technique using four fundamental variables (biomass, deciduousness, patchiness and soil type) and 65 vegetation plots to classify 7 types of vegetation in the Green Swamp: longleaf pine savanna; low-, medium-, and high-density low pocosin; high pocosin, pond pine woodland; and Atlantic white cedar (*Chamaecyparis thyoides*) forest (Kwasny 2001).

Summer 2001 will be used to ground-truth and correct the vegetation map. Personnel from IFSL will use the corrected vegetation map and gather additional fuel characteristics (such as crown-base height, crown bulk density, and fuel loading) to develop a fuels map and customized fuel models to enable the use of the Fire Area Simulator model (FARSITE). The FARSITE model is a deterministic fire growth simulator that applies fire behavior calculations to complex environmental conditions. It allows fuels, weather, and topography to vary spatially and temporally and produces maps of fire growth and behavior. It uses current and published fire behavior models to predict fire behavior (Finney 1998). After testing the accuracy of the vegetation maps and customized fuel models developed for the Green Swamp using recent fires, TNC hopes to better predict fire behavior in pocosin for the following: (1) to develop and test prescribed burn parameters using single source or single line ignition; (2) to simulate the effects of fuel mitigation along the preserve boundary that may be effective in reducing the threat of wildfires and increase the safety of prescribed burns on the preserve to plan fuel reduction projects; and (3) to predict potential fire growth and behavior of wildfires and effectiveness of suppression efforts. Moreover, TNC now has a permanent weather station in the Green Swamp that provides data for the national fire danger rating system through a cooperative agreement with NCDFR. Current and accurate fuel moisture and weather data will be key components for fire behavior predictions of future burns and in case of wildfires.

Conclusions

Managing ecosystems using prescribed fire as a landscape process requires burning across the boundaries of natural communities and ecosystems to create a mosaic of habitats and maintain the diversity inherent to those systems. But any management plan, especially one that involves a change of practices, must be based on good observation and research. Setting goals and carefully monitoring the vegetation response and effectiveness of the management activities are essential to developing science-based, ecological land management. In its Green Swamp preserve, TNC and its partners are putting these important principles into practice.

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The Role of Non-industrial Private Forest Lands in the Conservation of Southern Fire-dependent Wildlife

Christopher E. Moorman[1], *Peter T. Bromley*[2], *Mark A. Megalos*[3], *and David Drake*[4]

Abstract.—Although scientific support for fire as a land management tool has grown, non-industrial private forest (NIPF) landowners often fail to burn on their properties. These lands comprise approximately 70 percent of southern forests, making them critical to the long-term conservation of wildlife and plant species. Natural resource professionals must overcome key constraints to use of prescribed fire on NIPF lands if certain fire-dependent wildlife are to thrive on private forests. Results from two surveys suggest that fear of an escaped fire and related liability issues are the greatest landowner and manager concerns in North Carolina. Pro-fire media events and public education may be the best long-term solutions to increase southern NIPF landowner use of prescribed fire.

Introduction

Anthropogenic fire has a long history in the southern United States (Hudson 1982; Pyne 1982). Since their arrival over 10,000 years ago, Native American Indians burned southern forests and grasslands to drive game, improve grazing habitat, clear land, and reduce the chance of wildfire (Hudson 1982; Pyne 1982; Buckner 1989; MacCleery 1993). European immigrants readily adopted the Indians' woodsburning practices to improve range for cattle, reduce fuel loads, kill chiggers and ticks, increase visibility of snakes and large predators, and improve access (Stoddard 1962; Pyne 1982). Much of the Southeast burned every 1-6 years either at the hands of humans or from natural lightning ignitions (Frost 1998). These high frequency fires helped form the plant communities now present in the South (Christensen 1981; Buckner 1989; Frost 1998).

Because of its influence on plant communities, fire has played a central role in shaping the animal communities of the South as well. The value of fire as a tool to improve habitat for game species, such as the northern bobwhite (*Colinus virginianus*) has long been recognized (Stoddard 1935). However, fire protection policies, implemented in the 1920s, facilitated the decline of southern fire-dependent plant communities, such as the longleaf pine (*Pinus palustris*) forest, and the animals therein (Brockway and Lewis 1997; Brennan 1991; Engstrom et al. 1996).

Following declines in wildlife populations, much research was conducted, that highlighted the importance of fire to non-game species. Pine-dominated stands burned on short fire rotations (2-3 years) generally have a more diverse avian community than pine stands burned using less frequent fire (Wilson et al. 1995; Burger et al. 1998). Most bird species present in open pine-grasslands maintained by frequent fire are of equal or greater management concern than those that occur in the absence of fire (Sauer et al. 1996; Brennan et. al.1998; Burger et al. 1998). Declines in herpetofaunal abundance and diversity can occur following replacement of fire-adapted vegetation by fire-intolerant associations (Russell et al. 1999). Most reptile and amphibian species of conservation concern in the South, including gopher tortoise (*Gopherus polyphemus*), flatwoods salamander (*Ambystoma cingulatum*), indigo snake (*Drymarchon corais*) and pine barrens treefrog (*Hyla andersonii*), prefer habitats maintained by frequent fire (Means and Moler 1979; Brennan et al. 1998; Russell et al. 1999). Many southeastern small mammal species thrive in early- to midsuccessional habitats, many of which historically were created or maintained by periodic fire.

Although prescribed fire is recognized as a necessary habitat management tool for many non-game wildlife species, acreage burned in the South remains relatively stable and fire continues to be used on only a small fraction of NIPF lands (Brennan et al. 1998). Furthermore, NIPF lands that are burned may be done so only once in the length of a rotation or using fire frequencies longer than is desirable for maintenance of quality wildlife habitat (Brennan 1991; Drake 2000). Prescribed burning on NIPF lands is hampered by increasingly restrictive federal air quality standards, high equipment costs, liability risks, multiple ownership patterns or small tract sizes, financial limitations, and lack of landowner understanding of fire's value (Johnson 1984; Brennan 1991; Brennan et al. 1998; Izlar 2000). Public attitudes about fire have been greatly influenced by decades of fire prevention messages (e.g., Smokey Bear) emphasizing the destructiveness of wildfire (Gruell 1991). Although the anti-fire message excluded prescribed burning or natural fires, most citizens were unable to distinguish between good or bad fire (Little 1993). Media coverage has exacerbated the problem by dwelling on the sensational aspects of wildfire and doing little to educate the public on the benefits of prescribed burning (Gruell 1991).

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[1] Department of Forestry, North Carolina State University, Raleigh, NC 27695

[2] Department of Zoology, North Carolina State University, Raleigh, NC 27695

[3] North Carolina Division of Forest Resources, Raleigh, NC 27699

[4] Rutgers Cooperative Extension, New Brunswick, NJ 08901

Despite the barriers to prescribed burning on NIPF lands, there are many reasons why management of these forests must complement efforts to sustain fire-dependent ecosystems and their associated wildlife populations on public lands. Non-industrial private forests comprise a significant percentage (70%) of all timberlands in the South (Alig et al. 1990). Therefore, management efforts limited to public lands may not provide sufficient area to prevent fire-dependent species from declining or becoming extinct (Brennan et al. 1998). Populations of fire-dependent wildlife on public lands isolated by adjacent unburned private forests may experience negative effects (e.g., reduced access to resources, genetic deterioration, increased susceptibility to environmental catastrophes) commonly associated with habitat fragmentation (Harris 1984; Soulé 1987). Furthermore, private lands adjacent to public forests may have dangerously high fuel loads and neighboring landowners may harbor anti-fire sentiments. Either could eventually limit the ability of public land managers to burn. In response to relegating too large a conservation role to government (i.e., public lands), Aldo Leopold (1949) wrote, "An ethical obligation on the part of the private owner is the only visible remedy for these situations." Now, more than ever, Leopold's words ring true.

We use North Carolina as a case study to help clarify NIPF landowner attitudes pertaining to prescribed fire. We synthesize several surveys of North Carolina landowners and certified burners. Then, using literature accounts and survey results, we identify obstacles to burning in the South and the possible implications for southern fire-dependent wildlife. Finally, we discuss ways to increase both burning on NIPF lands and public support of prescribed burning as a tool to maintain and restore critical non-game wildlife habitats in the South.

North Carolina: a Case Study

Surveys of NIPF Landowners

Seventy-nine percent of North Carolina's forest occurs on NIPF lands (Alig et al. 1990), making it imperative that natural resource managers understand NIPF landowner attitudes related to forest management issues and what factors impact their land-use decisions. In North Carolina, 9% of landowners view wildlife (i.e., hunting and fishing) as a primary goal for owning and managing land, whereas an additional 25% manage their land for timber while protecting the environment or improving wildlife habitat and recreational opportunities (Megalos 2000). However, many North Carolina NIPF landowners neglect management of their forests because they own small tracts, they do not know where to start, or they do not rank forestry as a priority (Megalos 2000). Ninety-one percent of private landowners in North Carolina own tracts <50 acres, but these lands comprise only 28% of the total acres of private lands in North Carolina (Birch 1997).

Table 1.—Prescribed fire frequency on non-industrial private forest lands in the Sandhills of North Carolina (Drake 2000)

Frequency	Sample Size	Percent
Never	847	68.4
1-3 Years	847	9.4
4-6 Years	847	5.0
7-9 Years	847	1.8
^{3} 10 Years	847	2.8
Never, But Plan To In Future	847	13.1

A survey conducted in the Sandhills region of North Carolina identified reasons NIPF landowners fail to burn (Drake 2000). Of 873 landowners owning ^{3}30 acres, 81.5% had never burned their forests, and 68.4% never plan to burn (Table 1). Fire is a relatively popular management tool in the Sandhills region, meaning that even fewer landowners are likely to burn in other areas of North Carolina. Fear of escaped wildfire (41.5%) was the primary reason most landowners failed to burn, and 29% of those responding did not ever want fire on their property (Table 2). These reasons may be related to liability risks associated with prescribed fire and lack of understanding of the importance of fire in forest and wildlife management. The cost of burning was not included as a choice for landowners and may be an additional inhibitor to burning activities.

Survey of Certified Burners

In an effort to collect additional information on NIPF landowner attitudes related to prescribed fire, we mailed a 5-question survey to 292 burners certified through North Carolina's training program. Currently, most certified burners are employed by the North Carolina Division of Forest Resources, but some work for other state agencies (e.g., North Carolina Wildlife Resources Commission, Corps of Engineers) or conservation organizations (e.g., The Nature Conservancy). Many of these professionals are in contact with NIPF owners and were surmised to have excellent insights into the reasons landowners do or do not burn.

Sixty-four burners responded with partial or complete surveys. In the opinion of the burners, site preparation and fuel hazard reduction were the primary motivations for NIPF owners to burn (Table 3). Conversely, burning to increase biological diversity was an unlikely motivation ranked near the bottom of choices (Table 3). Certified burners ranked liability and smoke management concerns as the first and third most important reasons NIPF landowners fail or hesitate to burn (Table 4). Interestingly, the fear of losing control of a prescribed fire, which was the primary reason NIPF landowners in the Sandhills did not burn, was ranked second out of 11 choices by certified burners (Table 4).

Table 2.—Reasons non-industrial private forest landowners failed to burn their forests in the Sandhills Region, North Carolina (Drake 2000)

Reason for Not Burning	Sample Size	Percent
Worried About Fire Getting Out Of Control	626	41.5
Don't Want Fire On Property	627	29.0
Don't Know Where To Get Assistance	626	22.8
Don't Like the Looks	626	10.7
Developed Area Nearby	626	3.4
Other	626	24.0

Table 3.—Motivation for non-industrial private forest landowners to burn their woodlands in North Carolina, 2000 (scale ranges from -2 to 2: -2 = highly unlikely; -1 = unlikely; 0 = average; 1 = likely; 2 = very likely)

Reasons	Sample Size	Weighted Average (-2 - 2)
Site Preparation	61	1.21
Fuel Hazard Reduction	62	0.92
Hardwood Competition Control	62	0.63
Manage Game Animal Habitat	59	0.57
Aesthetics	62	-0.25
Increase Biological Diversity	62	-0.63
Pine Straw Production	61	-0.73

Certified burners recommended reducing landowner and burner liability, increased cost sharing, and more flexible smoke management guidelines as the government actions most likely to increase the use of prescribed fire on NIPF lands (Table 5). Development of a pro-fire media campaign, which likely would require less political activity than the top four approaches, was ranked fifth (Table 5). Thirty of the 44 burners that responded to a final open-ended question (What would you recommend in a more specific way to increase the use of prescribed fire on NIPF lands?) recommended either a pro-fire media campaign or landowner education programs to improve the general public's understanding and acceptance of prescribed fire.

Discussion

Implications for Wildlife Conservation

With liability and smoke management concerns identified as the primary barriers to burning, the future of prescribed burning, which likely will include equal or greater regulatory and legal restrictions, is uncertain. Increasingly restrictive air-quality guidelines in the future may further discourage NIPF landowner use of fire and may cut short recovery efforts for endangered species like the red-cockaded woodpecker (*Picoides borealis*) (Achtemeier et al. 1998).

Because ecosystems rarely conform to property lines, their maintenance or restoration generally requires coordination among multiple entities, including public land managers and private landowners (Brunson et al. 1996). Although habitat management on private lands is critical to the sustainability of all wildlife species, NIPF lands play an especially prominent role in maintenance of populations of rare and specialized wildlife species (e.g., fire-dependent species). Fifty percent of the country's threatened and endangered species are found only on private lands and an additional 20% spend approximately half of their time on private lands (Hunt 1997). Presently, the diversity of NIPF landowner backgrounds and objectives results in an extremely variable forest landscape across the South (Sheffield and Dickson 1998). Habitat specialists that are dependent upon fire-maintained habitats are especially vulnerable to habitat fragmentation (Hunt 1997). Therefore, the current pattern of constant change in forest condition from one landholding to the next most likely favors generalist wildlife species.

Brennan et al. (1998) predict continuing decline of prescribed burning on NIPF lands and refer to the impending isolation of fire-maintained habitats as an ongoing land use experiment. Such uncertainty does not bode well for fire-dependent wildlife. If suitable habitat

Table 4.—Reasons that best explain why non-industrial private forest landowners hesitate or fail to burn in North Carolina, 2000 (ranked from 1 = major reason to 11 = least important reason)

Reason	Sample	Average
Liability Concerns	61	3.23
Worried About Losing Control (Wildfire)	60	3.60
Smoke Management Concerns	61	3.74
Neighbor's Opposition	61	5.16
Cost	61	5.52
Conflict With Local Development	60	5.75
Limited Burning Days (Weather)	60	6.97
Doesn't Recognize Ecological Value	60	7.05
Doesn't Know Where To Get Help	61	7.56
Fire Line Considerations	61	7.80
Doesn't Like The Looks	61	8.84

Table 5.—Government actions most likely to increase the use of prescribed fire on non-industrial private forest lands in North Carolina, 2000 (ranked from 1 = major action to 7 = least important action)

Reasons	Sample Size	Average
Reduce Landowner Liability	61	3.75
Make Smoke/Fire Regulations More Flexible	61	3.90
Increase Cost Sharing	61	4.11
Reduce Certified Burner Liability	61	4.18
Develop A Pro-fire Media Campaign	61	4.48
Increase Landowner Education On Fire	61	4.62
Increase Direct Professional Assistance	61	4.67
More Proactive State Agencies	61	5.93

is present only as isolated pockets, dispersal by individual animals is limited and viable populations of many species may not be maintained over the long term (Noss 1991). Many fire-dependent wildlife species already are in decline or are listed as threatened or endangered (Brennan et al. 1998). As wildlife habitats are continually lost to population growth and urbanization, it will be imperative that NIPF landowners improve management of their forests for wildlife.

Possible Solutions

The problems confronting prescribed burners across the South are similar to those facing North Carolina's land management professionals. Average private landholding size continues to decline (Birch 1997) and forested tracts are progressively more isolated from one another by urban sprawl. Southern NIPF landowners increasingly are urban and absentee and have multiple management objectives (Boyce et al. 1986; Izlar 2000). Frequent changes in parcel ownership inhibit formation of productive relationships between local professionals and landowners and make aggressive marketing of state cost-share and assistance programs imperative. Smoke management regulations and tort liability add fuel to the problem by helping to deter NIPF landowners that otherwise might consider fire as a management tool (Brennan et al. 1998; Izlar 2000).

Southern resource professionals and landowners agree that liability concerns and increasingly restrictive air quality and smoke management guidelines are the greatest barriers to prescribed burning. Many southern states, including North Carolina, have passed legislation that in combination with certified burner training helps reduce burner liability (Achtemeier et al. 1998). Further reduction of burner liability may be unrealistic for accountability and legal reasons. Rather, legislators, resource managers, stakeholders and the Environmental Protection Agency should work jointly on legislation to exempt prescribed fire smoke emissions from air quality standards developed to reduce emissions from cars and industrial smokestacks. Additionally, continuing education and updates on new technologies (e.g., fire behavior models, weather prediction models and risk indices) should lessen the chance of negligent decisions by burners while widening the prescription window and increasing the number of acceptable burning days (Johnson 1984; Lavdas 1996; Achtemeier et al. 1998).

Divisions within and among agencies often arise between those who suppress fire and those who use it (Johnson 1984). To eliminate this dichotomy, the heads of state agencies must cooperatively lead a more proactive approach in marketing the benefits of prescribed fire to those that oppose it and budgeting sufficient resources to support prescribed burning efforts. Furthermore, the formation of a prescribed burning task force in every southern state would allow transfer of current information among resource professionals from different agencies and organizations while promoting constructive discussions on the current limitations (e.g., air quality guidelines) to use of prescribed fire on NIPF lands. Task force partners should include employees of the state agency in charge of fire suppression, members of the state wildlife agency including both game and non-game biologists, local Environmental Protection Agency (EPA) staff, legislators, and other stakeholders.

Adequate information pertaining to the cost effectiveness of prescribed burning is not available (Hesseln 2000). Recent research on the economic income forgone as a result of declining populations of fire-dependent species, such as the northern bobwhite (e.g., Burger et al. 1999), is a step in the right direction. Most research on the cost effectiveness of prescribed burning, however, has focused on the economic costs of burning (e.g., cost per acre) while failing to address social costs (e.g., costs derived from the inconvenience of smoke emissions), economic benefits (both market and nonmarket), and risk (Hesseln 2000). The long-term benefits of prescribed burning, including reduced risk of catastrophic wildfire, increased forage and habitat quality for wildlife, and enhanced biodiversity, may exceed short-term costs like reduced air quality, decreased aesthetics, risk of escape, and inconvenience from smoke (Hesseln 2000). However, without research and documentation of its financial efficacy, large-scale use of prescribed fire will be difficult to market to NIPF landowners and the general public. Such analyses would aid government in defining appropriate funding levels for support of prescribed burning (Hesseln 2000) and determining the merit of alternative management styles like fire suppression.

Natural resources historically have been undervalued (McNeely 1992). Similarly, there are significantly more disincentives than incentives for NIPF landowner use of fire. New financial incentives will be required to offset costs to landowners not currently using prescribed fire and to help state agencies, already short on money and manpower, meet increased demands for burning. Expanded cost-share programs (e.g., free firelines) can aid resource-limited landowners wishing to burn. Larger support budgets for professional assistance would provide the resources (e.g., on-site equipment, burning crews available 7 days/week and manpower to oversee permit approval) necessary to meet anticipated demands. If budgets are limited, cost-share programs and professional assistance could be prioritized to fund only burns that improve wildlife habitat or expand ecosystem restoration projects. Hazard reduction would be indirectly achieved on these lands. The creation of the longleaf pine ecosystem Conservation Priority Area under the Conservation Reserve Program (CRP) is an example of cost-share program that indirectly increases the demand for prescribed fire. Landowners receiving financial assistance for reestablishment of longleaf pine forests are more likely to use prescribed fire as a management tool, and, in the case of CRP landowners, will be required to use fire to remain eligible for program payments and benefits.

Forest management plans offer plan writers the opportunity to interact with landowners and discuss sustainable resource management alternatives, including the use of prescribed fire (Megalos 2000). Proponents of prescribed fire from state land grant universities or conservation organizations could work with consultants, state foresters, and other plan writers to encourage the use of prescribed fire. Government support of professional assistance programs, such as the Forest Stewardship Program, may help increase the number of NIPF landowners with management plans, while indirectly increasing the number of landowners burning their forests.

Increasing the use of prescribed fire on small NIPF lands or on forests within the urban-rural interface will be difficult. Owners of large forested tracts are most likely to use governmental cost-share money and public or private technical assistance (Franklin 1990). These landowners also are more likely to have a written management plan for their forests than owners of small parcels (Birch 1997). NIPF landowners with small acreages often have non-traditional management objectives (e.g., non-game management, aesthetic improvement) and generally own lands nearer to urban areas where neighbor opposition to fire is common. Owners of forested lands in the urban-rural interface may prefer a less-intensive, preservationist approach to land management.

Although small NIPF landholdings may contribute little to large-scale restoration of fire-dependent ecosystems, informing these landowners of the benefits of prescribed fire through extension education may improve the potential for future burning on public and larger private lands (Cortner et al. 1984; Taylor and Daniel 1984). In North Carolina, 42% of forest ownerships are between 20-100 acres (Birch 1987), making small tracts a necessary part of pro-fire programs. All voters and taxpayers, whether they own forestland or not, can influence fire-related policy, therefore they also should be targeted in outreach efforts. Workshops, bulletins, on-site visits, and land management demonstrations, traditionally administered by Cooperative Extension Service agents and specialists, should play a significant role in improving public understanding of fire-related issues. Many landowners agree to use of specific land management practices only after they have seen successful demonstrations (Brunson et al. 1996). Cooperative Extension provides a well-established link for information transfer among land grant universities, government agencies, and NIPF landowners across the South. Using modern technology, including the Worldwide Web and two-way video-teleconferences, foresters and wildlife biologists can reach a greater number of landowners more quickly and efficiently than ever (Bardon et al. 2000).

Public education pertaining to prescribed fire should focus on America's youth. Children generally are more open-minded than adults and are starved for new information on environmental issues. However, many live in urban areas and will be exposed to fire only through their classroom studies or what is seen on television. Proven environmental education programs offer a balanced, science-based source of information on the pros and cons of prescribed fire. Several existing environmental education curricula, including Project Wild and Project Learning Tree, contain activities related to fire ecology. Many conservation organizations and government agencies have completed or are in the process of developing similar programs. Natural resource professionals must aid environmental educators in distributing and teaching a proactive prescribed burning educational message in classrooms across the South.

Ultimately, television and news media must be used to balance the public's fear or dislike of fire with its positive effects. On-site telecasts, performed during a prescribed fire, could provide a dramatic background to capture viewer attention and allow burners the opportunity to discuss fire-related issues. Furthermore, an aggressive, pro-fire media campaign could help generate support for increased state agency burning budgets, exemptions to air quality guidelines, and legislation to reduce landowner and burner liability. Like firefighters, prescribed burners should be portrayed as heroes rather than villains (Murphy and Cole 1998). The old fire suppression message should be complemented with a new message relayed by new characters (e.g., Torchin' "Tom" Turkey and Burnin' "Bob"white) that promotes the use of prescribed fire for fuel hazard reduction, wildlife habitat improvement, and ecosystem restoration.

With the increasing popularity of herbicides, private industrial landowners continue to diminish the use of prescribed fire on their forestlands (Brennan et al. 1998). Herbicides applied for weed control and site preparation sufficiently fireproof plantations, and burning counters the value of industry's frequent fertilizations unless conducted ^{3}3 years after application (Gerhardt 2000). Similar to NIPF landowners, industry foresters avoid prescribed fire because of smoke management and liability concerns (Gerhardt 2000). More importantly, forest industry's approach to pine silviculture often is adopted by local NIPF landowners and forestry consultants. Herbicides often are portrayed as valuable wildlife habitat management tools and adequate substitutes for fire (Brennan et al. 1998). Rather, herbicides are a potential complement to fire when habitat management is a primary objective (Brennan et al. 1998; Brockway and Outcalt 2000). We believe forest industry should include the use of fire on NIPF lands as an option in their landowner assistance programs and maintain parcels of burned forests within corporate landholdings. Judicious use of fire can be proof positive of their commitment to plant and wildlife diversity.

As advances in the use of prescribed fire continue, resource managers must sell its comprehensive value to each and every NIPF landowner. Aldo Leopold (1949) criticized conservationists' tendency to apply economic value to all things endangered rather than justify conservation of species as a "biotic right". Similarly, we question threats of potential wildfire as the primary mode to encourage private landowners to use prescribed fire. Why not sell prescribed fire based on its ecological values in addition to its role in fuel hazard reduction? The more scientific information that can be provided about the consequences of using or suppressing fire, the better the general public's political and social decisions will be regarding the use of prescribed fire (Van Lear 2000).

Prescribed burning on NIPF lands could significantly aid restoration and maintenance of fire-dependent plant communities and the wildlife therein while concurrently reducing hazardous fuel loads. However, burning must be conducted frequently and during the right times of year. As ownership tenure shortens and as tract sizes lessens and becomes more fragmented, proper use of prescribed fire as a wildlife management tool will become increasingly difficult. Overcoming the multitude of barriers to use of fire will require a joint effort by all those that will benefit, including the general public. We feel large-scale, collaborative efforts to educate the pubic and use of new predictive technologies to reduce risk of escaped fire are the most effective and efficient means to increase burning on NIPF lands in the Southeast.

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Herbicides as an Alternative to Prescribed Burning for Achieving Wildlife Management Objectives

T. Bently Wigley[1], *Karl V. Miller*[2], *David S. deCalesta*[3], *and Mark W. Thomas*[4]

Abstract.—Prescribed burning is used for many silvicultural and wildlife management objectives. However, the use of prescribed burning can be constrained due to difficulties in obtaining burning permits, concerns about liability, potential effects of scorch on growth and survival of crop trees, its sometimes ineffective results, limited burning days, and the costs of applying, controlling, and monitoring burns. For some landowners, herbicides offer a cost-effective alternative to prescribed burning for manipulating plant communities and wildlife habitat, especially when the boundaries of application are closely defined and the focus is on individual habitat components. Although the ecological effects of fire and herbicides sometimes differ, when used alone or with other management practices herbicides offer an opportunity to meet many wildlife management objectives. In this paper, we discuss and provide examples of wildlife management objectives that have been met by using herbicides, and factors that should be evaluated when considering use of either prescribed burning or herbicides.

Introduction

Wildlife habitat is "an area with the combination of resources (like food, cover, water) and environmental conditions (temperature, precipitation, presence or absence of predators and competitors) that promotes occupancy by individuals of a given species (or population) and allows those individuals to survive and reproduce" (Morrison et al. 1992:11). Achieving management objectives for single wildlife species, or communities of wildlife species often involves manipulating in space and time the structure, composition, and distribution of plant communities and special habitat features such as snags, down and dead wood, and mast-producting vegetation.

Fire has long been used for managing plant communities. Native Americans burned forest land periodically to improve game habitat, facilitate travel, reduce insect pests, remove cover for potential enemies, and enhance native food production (MacCleery 1992, Day 1953). Early European settlers used fire to improve habitat for livestock and game species such as white-tailed deer (*Odocoileus virginianus*), wild turkey (*Meleagris gallopavo*), and northern bobwhites (*Colinus virginianus*) (Komarek 1981). Currently, foresters and wildlife managers prescribe fire to reduce fuels, prepare sites for natural or artificial regeneration, control competing vegetation in mid-rotation stands, control certain insects and diseases, enhance development of forage resources, obtain desired structural characteristics (e.g., development/promotion of herbaceous and shrub layers), create specialized habitat components (e.g., snags and logs), and restore desired plant species composition in some ecosystems, e.g., longleaf pine (*Pinus palustris*).

As recently as 20 years ago, prescribed burning was used extensively to manage plant communities on private lands (Mobley and Balmer 1981), which represents the majority of lands in the United States (USDA Forest Service 2000). However, in many states the use of prescribed burning appears to have been relatively stable or slightly declining over the past 20 years, although data related to these trends are limited. In a survey of southern state forestry agencies by the Georgia Forestry Commission, 6 responding states indicated that the area burned over the past 2 decades has remained relatively stable and 2 states reported a significant decline in area burned (R. Ferris, Georgia Forestry Commission, personal communication). Trends in states not responding are unknown. Data from South Carolina provides an example of a state where the area burned annually has been slightly declining over the past 20 years (Figure 1). In areas where use of prescribed burning is constrained or declining, managers have begun to search for alternative technologies to achieve wildlife management objectives.

During the latter half of the 20th century, herbicides emerged as a tool for manipulating plant communities. Herbicide products (generally the active ingredient and one or more surfactants mixed in water) are used extensively to manipulate the species composition and structure of vegetation in agriculture, along roads and utility rights-of-way, in urban settings, and in forest management (Walstad and Kuch 1987, Brennan et al. 1998). However, data describing trends in herbicide use in forested ecosystems in the United States are limited.

The recent registration of more selective herbicides increases the potential to use herbicides for achieving wildlife management objectives, especially when these objectives cannot be achieved through prescribed

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[1] NCASI, Clemson SC 29634

[2] Warnell School of Forest Resources, University of Georgia, Athens, GA 30602

[3] Northeastern Research Station, USDA Forest Service, Irvine, PA 16329

[4] Wildlife Intergration, Birmingham, AL 35244

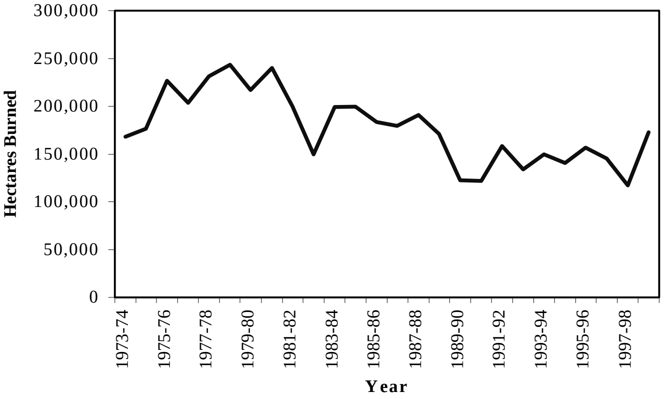

Figure 1.—Area treated with prescribed fire in South Carolina during 1973-1999 (source, South Carolina Forestry Commission annual reports).

burning or in forest systems where fire is not a natural ecological force. In this paper, we discuss the use of herbicides to address wildlife-related objectives within forested ecosystems, with an emphasis on the eastern United States. We will describe herbicides commonly used in forest management, silvicultural objectives for which they are used, habitat components affected, and wildlife objectives that can be met with their use. We also will discuss considerations for determining which tool to use. Our manuscript benefitted from reviews by R. A. Lautenschlager and D. H. Van Lear.

Ecological Functions of Fire

Can the judicious use of herbicides lead to conditions similar to those created by fire? The answer depends upon the specific ecological response in question. Fire has myriad effects in forested ecosystems. Fire influences plant and animal species richness, plant reproduction and development, insect outbreak and disease cycles, wildlife habitat relationships, soil functions, and nutrient cycling (SNEP Science Team and Special Consultants 1996). The ecological effects of fire (Figure 2) are complex, interrelated, and sometimes undesirable when fire is intense or occurs across large areas (Agee 1993). Pyne et al. (1996), based on information in Wright and Heinselman (1973), suggested that depending upon intensity fire may:

- Trigger the release and germination of seeds in some plant species;
- Stimulate flowering and fruiting of some shrubs and herbs;
- Alter seedbeds by removing litter and humus and creating bare soil;
- Stimulate vegetative reproduction of woody and herbaceous species through overstory reduction;
- Temporarily reduce competition for moisture, nutrients, and light, thereby favoring some species;
- Selectively eliminate part of a plant community;
- Influence community composition and successional stage; and
- Regulate susceptibility of forests to blowdowns.

Fire has countless other ecological effects some of which depend upon the ecosystem in which it occurs. Fire may kill or injure above- and below-ground portions of plants, volatize nitrogen, improve conditions for nitrogen mineralization, cause elements/nutrients to become more available for uptake by plants, and dramatically change micro-climates (Wright and Heinselman 1973). In oak ecosystems, fire creates favorable conditions for acorn caching by squirrels (*Sciurus* spp.) and blue jays (*Cyanocitta cristata*), reduces populations of insects that prey on acorns and young oak seedlings, xerifies mesic sites through consumption of surface organic matter and exposure of the soil to greater solar radiation, and reduces understory and midstory competition from fire-intolerant species (Van Lear and Watt 1993). Fire scarifies the seed coat of some plants and enhances their germination, and reduces debris loading following natural disturbance or harvesting.

Clearly, use of herbicides also results in some of these ecological effects. Herbicides can injure or kill the above-ground portion of plants, selectively eliminate part of a plant community, influence community composition and successional stage, and temporarily reduce competition among plants for resources. In such cases, herbicides may provide an appropriate substitute for prescribed burning. However, herbicides cannot perform every ecological function of fire. For instance, herbicides cannot directly and immediately alter a seedbed by removing litter and humus and creating bare soil, although herbicides can contribute to this indirectly over time. Herbicides cannot scarify leguminous seeds to enhance germination or stimulate seed release in plants such as jack pine (*Pinus banksiana*).

Herbicides may be more effective at eliciting some ecological effects if used in combination with other management tools. For example, mechanical site preparation could be used in combination with herbicides to remove litter and humus and create bare soil. Herbicides and fire already are commonly used in combination for site preparation to reduce debris loading and control competing competition. However, approaches for combining herbicides and other tools to meet wildlife and ecological objectives need more thorough investigation.

Herbicide Use in Forestry

Wildlife habitat management is commonly achieved in conjunction with or as a corollary of other land management activities such as forestry. Often, the decision of whether to use fire or herbicides for wildlife

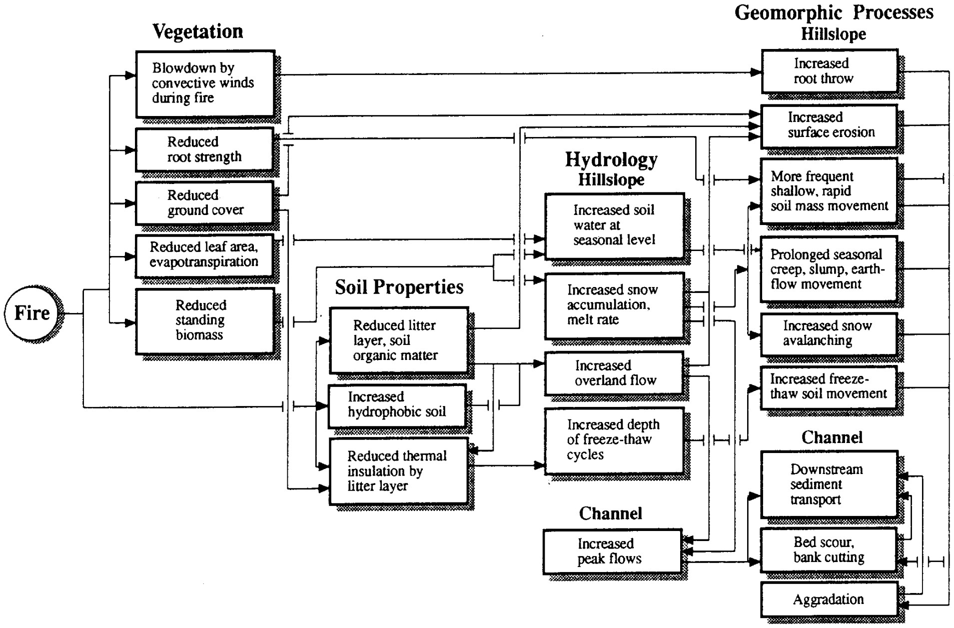

Figure 2.—The effects of fire on vegetation, soils, hydrology, and geomorphic processes (from Swanson 1981 and Agee 1993). Excerpted from *Fire Ecology of Pacific Northwest Forests* by James K. Agee. Copyright © 1993 by James K. Agee. Reprinted by permission of Island Press, Washington, DC and Covelo, CA.

management in forested ecosystems depends upon which tool is most effective at achieving other landowner objectives (e.g., a forestry objective). Because herbicides are increasingly a preferred tool for achieving forestry objectives, we will briefly describe forestry-related uses of herbicides. Each of these forestry-related uses represents opportunities for biologists to interact with foresters and discuss modifications to herbicide prescriptions that would also achieve wildlife management objectives.

Herbicides are used in forestry for site preparation, release of crop trees from competition with herbaceous and non-commercial woody plants, and timber stand improvement (Lautenschlager 2000). The reduction of competing vegetation can significantly increase tree growth well into mid-rotation (e.g., Zutter and Miller 1998), and controlling both woody and herbaceous vegetation provides the greatest increase in tree growth (Figure 3). Herbicide applications typically are tailored according to soils, structure and composition of the plant community, and management objectives. Table 1 provides an overview of herbicides commonly used in forest management.

Depending upon topography and soil conditions, site preparation may be accomplished using herbicides alone or in combination with mechanical methods or fire. When applied for site preparation, herbicides generally are broadcast. Thus, using herbicides alone for site preparation (especially when they are aerially broadcast) generally results in minimal soil disturbance and erosion potential.

To control herbaceous vegetation, herbicides often are broadcast or applied in bands or spots during the first year or two following stand establishment. Some herbicides, such as sulfometuron can be sprayed over the top of the seedlings of selected tree species (e.g., southern pines) without adversely affecting their growth. Following stand establishment and through mid-rotation, herbicides are commonly used to release crop trees from the influence of competing vegetation. Sometimes herbicide applications for this purpose follow thinnings or precede applications of fertilizer.

As an intermediate treatment (timber stand improvement), hebicides often are applied to individual woody stems in the midstory and overstory to improve the composition, structure, condition, and growth of the stand. Herbicides can be applied during much of the year to individual woody stems through injection (herbicide applied to a wound in the tree bole), basal spraying (herbicide sprayed at the base of the tree close to the ground), or soil treatment (herbicide applied to ground), although there may be some seasonal constraints on these treatments. Treatment of individual stems is labor intensive, but the ability to do so provides significant opportunities for selective habitat enhancement without impacting the entire plant community.

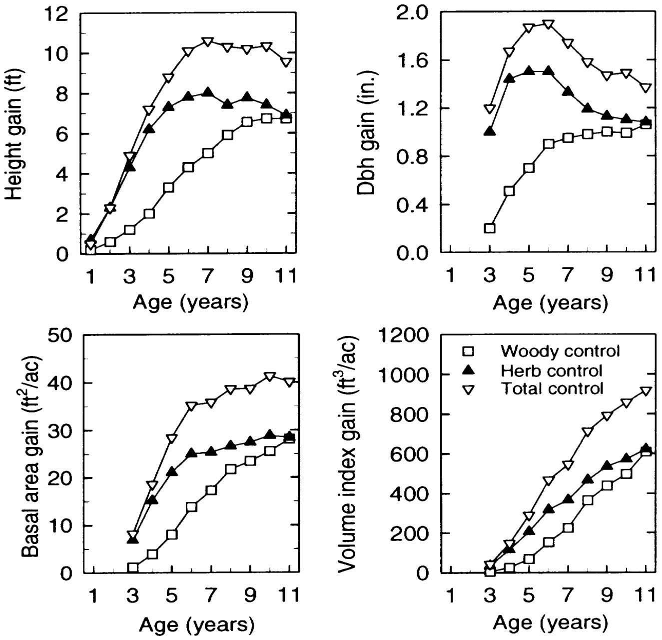

Figure 3.—Gains in average loblolly pine height, dbh, stand basal area, and stand volume index over no-treatment control through 11 growing seasons by vegetation control method (from Zutter and Miller 1998). Reprinted from the Southern J. Appl. For. 22[2]:93 published by the Society of American Foresters, 5400 Grosvenor Lane, Bethesda, MD 20814-2198. Not for further reproduction.

The various herbicides registered for forestry use typically affect different plant species and species groups. Some herbicides such as glyphosate are broad spectrum and affect virtually all plant species, although timing and application rates may alter selectivity of many herbicides. Other herbicides are more selective and affect only certain species or plant growth forms (Table 2). For example, metsulfuron is highly effective for controlling plants in the genus *Rubus*. In contrast, legumes and *Rubus* spp. generally are tolerant to imazapyr (Table 3). Fluazifop-P and sethoxdim are grass-specific chemicals and have little impact on broadleaf species, while triclopyr has little effect on grasses and sedges. Sulfometuron methyl (as Oust®) has been used in northern hardwood forests to control hay-scented (*Dennstaedtia punctilobula*) and New York (*Thelypteris noveborancensis*) ferns but was found to have no effect on woody plants (Horsley 1988a).

Sometimes different configurations of the same herbicide have different effects, due either to differences in the nature of the active ingredient (i.e., ester or amine) or additives (e.g., type of surfactant). For example, Miller and Mitchell (1990) found that applications of triclopyr in the form of Garlon® resulted in 40-80 percent mortality in dogwoods (*Cornus* spp.), while applications of triclopyr in the form of Pathfinder® resulted in >80 percent mortality of the same species. This selectivity enhances a manager's ability to manipulate plant communities. Of course, because of this differing selectivity, foresters sometimes mix two or more herbicides in the same tank to enhance the number of species controlled during an application. However, some herbicides are not compatible in tank-mixes and the number of species controlled by such mixtures may actually decrease (Ezell 1998).

Using Herbicides to Meet Wildlife Management Objectives

Although the ecological effects of herbicides and fire sometimes differ, herbicides can be used to meet many wildlife management objectives related to plant species composition and structure, special habitat features (e.g., snags, down wood), and the temporal and spatial distribution of selected habitat components. In fact, herbicides are more effective than fire for achieving some wildlife management objectives and can perform some functions that fire cannot. Numerous studies have evaluated the potential of using herbicides for specific wildlife management objectives (Table 4). In reviewing many of these studies, Lautenschlager et al. (1995) suggested that, by choosing appropriately (active ingredient, time of application, application technique), herbicides can be used to: (1) reduce densities of invading non-native plants (restoring native populations and associated wildlife); (2) create snags, dead and down woody material, and "drumming logs" in early or later successional stands (providing "old growth" characteristics); (3) create small, intermediate, or large early-successional openings within older vegetation types; (4) change shrub-dominated areas to earlier successional grassy, or herb/grass-dominated communities; (5) favor male aspen clones; (6) release patches or expanses of conifers; and (7) keep woody and herbaceous "browse" within reach of browsing animals.

Managing Vegetative Species Composition and Structure

By using newer, more selective herbicides or regulating time of application, managers can manipulate understory plant species composition and structure. For example, dense mats of hay-scented fern and New York fern can interfere with development of woody seedlings and the shrub layer in northern hardwood forests (DeGraaf et al. 1992, Horsley 1988b). This reduces food resources (fruits from shrubs, woody browse from seedlings) and vertical structure (shrub and midstory layers) for many wildlife species, especially songbirds. Applying herbicides during late summer and early fall generally will control ferns and result in little if any damage to desirable woody seedlings or to spring ephemeral herbs, which already have completed their annual reproductive cycles and senesced (Ristau and Horsley 1999)

Table 1.—Characteristics of common silvicultural herbicides and herbicide formulations

Herbicide	Trade Name	Site Preparation	Conifer Release	Herbaceous weed control	Tree Injection	Cut Stump Application	Hardwood Weed Control	Basal Bark Applications	Activity	Behavior in soil	Toxicity
Dicamba	Vanquish	X			X	X			Foliar; Soil	Weakly adsorbed	Low
Fluazifop-P	Fusilade DX			X					Foliar	Readily adsorbed, low mobility	Very low
Glyphosate	Accord, Roundup	X	X	X	X	X			Foliar	Rapidly adsorbed to soil	Very low
Hexazinone	Pronone 10G, 25G, and MG	X	X	X					Soil	Relatively mobile	Low
Hexazinone	Velpar L	X	X	X	X				Soil; Foliar	Relatively mobile	Low
Hexazinone	Velpar ULW	X	X						Soil; Foliar	Relatively mobile	Low
Imazapyr	Arsenal AC	X	X	X	X	X			Foliar; Soil	Weakly adsorbed to soil	Very low
Imazapyr	Chopper	X	X		X	X		X	Foliar; Soil	Weakly adsorbed to soil	Very low
Imazaquin	Scepter			X			X		Foliar: Soil	Weakly adsorbed in high pH soils	Verylow
Metsulfuron	Escort			X					Foliar; Soil	Moderately mobile	Verylow
Pendimethalin	Pendulum			X			X		Soil	Strongly adsorbed to soil	Very low
Picloram	Tordon K	X							Foliar	Weakly adsorbed by clays	Low
Picloram + 2,4-D	Pathway				X	X			N/A	Weakly adsorbed by clays	Low
Picloram + 2,4-D	Tordon 101M	X			X	X			Foliar	Weakly adsorbed by clays	Low

Continued

Herbicide	Trade Name	Site Preparation	Conifer Release	Herbaceous weed control	Tree Injection	Cut Stump Application	Hardwood Weed Control	Basal Bark Applications	Activity	Behavior in soil	Toxicity
Sethoxdim	Vantage			X					Foliar	Adsorption varies with organic material	Very low
Sulfometuron	Oust			X			X		Foliar; Soil	Not tightly bound (especially high pH)	Very low
Triclopyr	Garlon 3A	X	X		X	X			Foliar	Not tightly bound	Low
Triclopyr	Pathfinder II					X		X	N/A	Not tightly bound	Low
Triclopyr	Garlon 4	X	X			X		X	Foliar	Not tightly	Low

Table 2.—Susceptibility[1] of selected woody plant species to forestry herbicides. Data obtained from product labels and Miller and Mitchell (1990)

		Woody plant species																																			
Herbicide	Trade Name	*Abies* sp.	*Acer* spp.	*Alnus* spp.	*Amelanchier* spp.	*Betula* spp.	*Carpinus* sp.	*Carya* spp.	*Chamaecyparis* sp.	*Corylus* spp.	*Cornus* spp.	*Crataegus* spp.	*Diospyros* sp.	*Fraxinus* spp.	*Juglans* sp.	*Juniperus* spp.	*Larix* spp.	*Liquidambar* sp.	*Liriodendron* sp.	*Lonicera* spp.	*Morus* spp.	*Oxydendrum* sp.	*Picea* spp.	*Pinus* spp.	*Populus* spp.	*Prunus* spp.	*Quercus* spp.	*Rhus* spp.	*Ribes* sp.	*Rosa* spp.	*Rubus* spp.	*Salix* spp.	*Sambucus* sp.	*Toxicodendron* spp.	*Tsuga* sp.	*Ulmus* spp.	*Vitis* spp
Dicamba	Vanquish; Banvel		I	S	S	S	S	S			S	S	T	S				T		S		I	S	S	S	I	I	S		S	S	S		S	S	S	S
Glyphosate	Accord; Roundup	T	I	S		S		T		S	S	S	I	T				S	I	S		I	T	T	S	I	S	I		S	S	S	S	S	T	I	
Hexazinone	Pronone 10G, 25G, and MG	T	I			S		T		S	T	S	T	I				S	T	S		T	T	T	S	I	S	T			S	S				S	
Hexazinone	Velpar L and Velpar ULW	T	I			S		I		S	I	S	T	I				I	T	I		T	T	T	S	I	S	I			S	S				S	S
Imazapyr	Arsenal AC		S					I			I	T	S	S				S	S	S	S		T	T	S	I	S	S		S	S	S		S		T	S
Imazapyr	Chopper RTU		S					S			S	S		S											S	S	S	S		S		S					
Metsulfuron	Escort	S	S	S								S		S		S			S	S	S		S		S	S	S			S	S	S				S	S
Pendimethalin	Pendulum	T	T	T		T			T		T	T		T	T	T			T			T	T	T	T	T	T					T			T	T	
Picloram	Tordon K	S	S					I			I	S	I			S		I					S	I	S	I	S	S			S	S			S		
Picloram + 2,4-D	Pathway	S	S	S	S	S	S	S			S	S		S		S									S	S	S									S	
Picloram + 2,4-D	Tordon 101M	S	I	S		S		S					S					I	S	S		S	S	I	S	S	S	S			S	S			S	I	
Sethoxdim	Vantage; Poast	T	T			T			T		T			T	T			T					T	T	T	T	T					T			T		
Sulfometuron	Oust		R											T			R						R	R	T		R	I			I						
Triclopyr	Garlon 3A	T	I	S		S		I		S	I	T		S				I	S		S			I	S	I	I	S			S	S	S	S		I	
Triclopyr	Garlon 4		I	S		S		S		S	I	T	T	I				I	S		S		I	I	S	I	I	T			S	S	S	S		S	
Triclopyr	Pathfinder II		S	S		S		S		S	S		S	S		S	S	S	S				I	I	S	I	S	I			S	S		S		I	

[1] S=Susceptible species (>80 percent control); I=Marginally or variably controlled species (40-80 percent control); T=Tolerant species (<40 percent control); blank = No data available).

Table 3.—Plant species that are tolerant to imazapyr or that commonly recolonize a site following an application of imazapyr (from American Cyanamid Company 1999)

Tolerant		Recolonize	
Scientific name	Common name	Scientific name	Common name
Amorpha fruticosa	Indigo bush	*Amaranthus hybridus*	Pigweed
Amphicarpa bracteata	Hog peanut	*Ambrosia artemisifolia*	Common ragweed
Apios americana	Ground nut	*Ambrosia trifida*	Giant ragweed
Cassia fasciculata	Partridge pea	*Andropogon* spp.	Broomsedges
Cassia nictitans	Small partridge pea	*Bidens spp.*	Beggar ticks
Centrosema virginianum	Butterfly pea	*Callicarpa americana*	American beautyberry
Cercis canadensis	Redbud	*Campsis radicans*	Trumpet vine
Clitoria mariana	Butterfly pea	*Ceanothus americanus*	New Jersey tea
Desmodium nudiflorum	Beggarweed	*Chenopodium album*	Lambsquarters
Desmodium rotundifolium	Beggarweed	*Croton capitatus*	Wooly croton
Desmodium tortuosum	Florida beggarweed	*Croton glandulosus*	Dove weed
Ephrosia virginiana	Goats rue	*Cuscuta gronovii*	Lovevine
Galactia volubilis	Erect milk pea	*Diodia teres*	Poor-joe
Indigofera caroliniana	Wild indigo	*Epilobium angustifolium*	Fireweed
Lespedeza bicolor	Bicolor lespedeza	*Euphorbia corollata*	Flowering spurge
Lespedeza capitata	Roundhead lespedeza	*Geranium carolinianum*	Wild geranium
Lespedeza hirta	Hairy lespedeza	*Ipomoea purpurea*	Morningglory
Lespedeza intermedia	Wand lespedeza	*Mollugo verticillata*	Carpet-weed
Lespedeza japonica	Japonica lespedeza	*Oenothera biennis*	Evening primrose
Lespedeza procumbens	Prostrate lespedeza	*Oxalis stricta*	Yellow wood sorrel
Lespedeza striata	Common lespedeza	*Panicum* spp.	Panic grasses
Lespedeza thunburgii	Thunburg lespedeza	*Parthenocissus quinquefolia*	Virginia creeper
Psoralea psoralioides	Samson snakeroot	*Passiflora incarnata*	Maypop
Rhynchosia reniformis	Dollar weed	*Physalis virginiana*	Ground cherry
Rhynchosia tomentosa	Hairy rhynchosia	*Phytolacca americana*	Pokeweed
Robinia pseudo-acacia	Black locust	*Polygonum pennsylvanicum*	Pennsylvania smartweed
Rubus argutus	Blackberry	*Rhus copallina*	Winged sumac
Rubus trivialis	Dewberry	*Rhus glabra*	Smooth sumac
Schrankia microphylla	Sensitive briar	*Rhus radicans*	Poison ivy
Sesbania macrocarpa	Sesbania	*Richardia scabra*	Florida purslane
Strophostyles helvola	Milk pea	*Rumex hastatulus*	Sheep-sorrel
Strophostyles umbellata	Trailing wild bean	*Smilax bona-nox*	Greenbrier
Stylosanthes biflora	Pencil flower	*Trichostema dichotomum*	Blue curls
Tephrosia spicata	Spike tephrosia	*Viola* spp.	Violets
Vicia dasycarpa	Narrowlearf vetch	*Vitis rotundifolia*	Muscadine grape
Vigna suteola	Wild pea		

Table 4.—Examples of wildlife habitat objectives achieved or resulting through use of herbicides

Objective	Location	Citation
Reduce live emergent vegetation in wetlands	North Dakota South Dakota	Blixt (1993) Solberg and Higgins (1993)
Reduce abundance of parasites in small mammals	Oklahoma	Boggs et al. (1991)
Increase selected wildlife foods and cover, and habitat interspersion	Pennsylvania	Bramble and Byrnes (1983)
Create snags to accelerate development of old-growth characteristics	Oregon	Cole (1996)
Manage hardwood midstory in red-cockaded woodpeckers (*Picoides borealis*) cluster areas	Texas Georgia	Conner (1989) Jones (1992)
Reduce habitat suitability for northern pocket gophers (*Thomomys talpoides*) to control damage to lodgepole pine (*Pinus contorta*) seedlings	Unknown	Engeman et al. (1997)
Establish food plots	Wisconsin	Hamilton and Buckholtz (1953)
Control undesirable emergent vegetation and promote waterfowl food plants in impoundments	Georgia	Wood et al. (1996)
Manipulate lesser prairie chicken (*Tympanuchus pallidicinctus*) habitat	Texas	Doerr and Guthery (1983)
Improve elk (*Cervus elaphus canadensis*) and mule deer (*Odocoileus hemionus*) range	Colorado	Kufeld (1977)
Restore herbaceous understory in pine stands managed for northern bobwhite (*Colinus virginianus*)	Florida	Welch (2000)
Create habitat for cavity-nesting songbirds	Kentucky	McComb and Rumsey (1983)
Provide openings and increase deer (*Odocoileus virginianus*) forage	Oklahoma	Thompson et al. (1991)

Directed application of herbicides also can be used to suppress some woody species from the shrub and midstory layers, thus promoting growth and development of species with more desirable structural features. For example, shrub-nesting songbirds prefer the finer and multiple-branching twigs produced by species such as American beech (*Fagus grandiflora*) and birches (*Betula* spp.) to the more simplified branching of larger twigs represented by striped maple (*Acer pensylvanicum*) (D. S. deCalesta, USDA Forest Service, unpublished data). Herbicides can be used to selectively reduce the abundance of striped maple in the shrub and midstory, which competes with species such as beech and birches. Such application can be expensive, however, and uneconomical when treatment levels exceed 400 stems per acre (R. D. Nyland, State University of New York, School of Environmental Science and Forestry, personal communication).

Annual or biennial prescribed burning during the dormant season has been unable to effectively control understory hardwood invasion in some open pine forests managed for red-cockaded woodpeckers (*Picoides borealis*) and northern bobwhite (*Colinus virginianus*). Welch (2000) reported that a one-time application of imazapyr alone or combined with prescribed burning could significantly reduce hardwood invasion without adversely impacting habitat conditions or food production for northern bobwhites and allow future management with prescribed fire during the growing season. This strategy sometimes is used by federal agencies (Ralph Costa, U.S. Fish and Wildlife Service, personal communication).

Herbicides can be used in conjunction with regeneration techniques, such as a shelterwood harvest, to alter overstory species composition and structure through management of advanced regeneration. A combined shelterwood harvest and herbicide application increases sunlight to the forest floor and stimulates germination and growth of seeds thrown by the overstory. If conditions are appropriate, prescribed burning also can be used for similar purposes (Brose et al. 1999).

Managing Special Habitat Features

The availability of snags and coarse woody debris is a key factor influencing the abundance and composition of wildlife communities. Snags and down wood are created by a number of factors, including shearing winds, rot associated with insect and disease attack, lightning strikes, and wild fire. However, these natural processes produce somewhat variable and unpredictable results in terms of the abundance and characteristics of created snags (e.g., species composition, dbh, height,). Light prescribed burnings may not cause enough damage to the cambium to lead to tree mortality and create snags, especially for tree species that are resistant to fire-induced mortality. Thus, snags and down wood may not be created from all species.

Herbicides have been used to create snags for a variety of wildlife-related purposes (Conner et al. 1983, Bull and Partridge 1986). Because they can be applied selectively to individual trees, herbicides can be used to regulate the species composition, dbh, and height of snags and resulting logs. Snags created with some herbicides (e.g., 2,4-D) may decay more rapidly than snags created through other means such as girdling (Conner et al. 1983; Bull and Partridge 1986). However, ongoing research in Oregon (Michael Newton, Oregon State University, personal communication) suggests that the life span of snags created through mechanical means (e.g., girdling, topping) and herbicides such as MSMA and triclopyr can be very similar.

Managing Spatial and Temporal Arrangement of Habitat

Herbicides can be used to manage the spatial and temporal availability of habitat, a prime determinant of the diversity and productivity of wildlife communities (Morrison et al. 1992). For example, herbicides can be used to create snags and down wood where desired within the landscape and in a variety of seral stages. Managers can use herbicides to retain and regulate the distribution of conifers in riparian ecosystems in order to provide nesting and foraging habitat for bird species such as blackburnian warbler (*Dendroica fusca*), Swainson's thrush (*Catharus ustulatus*), and Acadian flycatcher (*Empidonax virescens*). Herbicides can be used in selected locations to produce patches of early-successional habitats and change overstory species composition. Overstory species composition can be changed directly by killing undesired overstory trees or indirectly and over a long period of time by altering species composition of advanced regeneration as previously discussed.

Considerations When Choosing Between Fire and Herbicides

As vegetation management tools, herbicides and fire each have a unique set of advantages and disadvantages. The decision to use fire or herbicides is complex and involves many variables. We recommend that biologists and managers consider the following factors when deciding when and where to apply these tools.

Effectiveness

Obviously, managers should weigh the relative capabilities of prescribed burning and herbicides to achieve desired vegetative conditions. For some conditions, prescribed burning is most appropriate (e.g., promotion of fire-adapted understory vegetation). Sometimes, however, herbicides can be equally or more effective at eliciting desired vegetative responses. For example, herbicides are a unique and effective tool for accelerating the development of late-successional habitat, specific old-growth components (e.g., large snags and logs, large live trees of specified species composition), and associated wildlife species (e.g., Cole 1996). This can be accomplished by turning some live overstory trees first into snags of desired species, dbh, and spatial distribution, and later into logs when they fall. Herbicides are a unique tool for controlling populations of some non-native species. For example, Grilz and Romo (1995) found that smooth brome (*Bromus inermis*) was most effectively controlled by spring burning combined with glyphosate applications. Herbicides are particularly well suited for regulating plant communities in early successional habitats where regenerating trees would be damaged by fire.

Historical Disturbance Regime

In selecting whether to use prescribed burning or herbicides, managers also should consider disturbance regimes of the ecosystem being managed. Generally, prescribed burning is most appropriate in fire-associated or fire-dependent ecosystems such as pine and oak ecosystems that historically were disturbed on a regular basis by non-lethal understory fires (Abrams 1992; Waldrop and Van Lear 1989). However, even in these forest types fire was not the only form of historical disturbance. For example, in southern pine forests, hurricanes, ice storms, and southern pine beetles (*Dendroctonus frontalis*) also helped shape forest structure, species composition, and habitat for species such as red-cockaded woodpeckers (*Picoides borealis*) (Coulson et al. 1995; Hooper and MacAdie 1995; Conner and Rudolph 1995). These disturbance factors created important habitat features (e.g., snags, dead down wood) not readily created through low-intensity fires with short return intervals.

In some forest ecosystems, historical fires affected small areas, were infrequent, or occured primarily as stand-replacing fires or mixed and variable fires (Brown 1994, Runkle 1985). Historical return intervals of fire in some forest ecosystems in North America are estimated to be as long as 500-1,000 years, e.g., northern New England (800 years), upper elevation conifer forests in eastern Canada (1,000 years), coastal redwood forests in California (500-600 years) (Oliver and Larson 1990). In such situations, prescribed burning may not be the most appropriate tool for achieving habitat objectives and could cause damage to trees that are not fire-adapted. For example, in the Northeast, management for species such as chestnut-sided warblers (*Dendroica pensylvanica*), bluebirds (*Sialia sialis*), and bobolinks (*Dolichonyx oryzivorus*) requires development of early-successional habitat (Braile 2000). However, before timber harvest can be used to create these habitats, the density of ferns, grasses, blackberries (*Rubus* spp.), and undesirable woody species often must be reduced to allow sufficient stocking of advanced regeneration of desirable (ecologically and commercially) tree species. This objective may best be achieved using herbicides, particularly in ecosystems where fire is not the dominant source of disturbance. Where oaks are not fire-adapted, managers may choose to promote oak regeneration by top-clipping oak seedlings and treating the remaining vegetation with a herbicide such as glyphosate. The top-clipped oak seedlings, which will not have absorbed the herbicide, will sprout and grow vigorously in the absence of competing vegetation (Wright et al. 1985).

Risk to Other Resources

Managers sometimes choose to use herbicides because fire can damage other resources. For example, extremely hot fires can alter the physical properties of soils, accelerate erosion rates, volatize nutrients, and slow successional recovery (Pyne et al. 1996, Lautenschlager et al. 1998). Crown scorch can cause mortality and loss of diameter and height growth in crop trees (Waldrop and Van Lear 1984). Johansen and Wade (1987) reported that even slightly scorched trees showed a 15 percent loss of radial growth. Because managed forests represent a signficant financial investment, many landowners are hesitant to risk such losses.

Administrative Considerations

Herbicides may be an appropriate tool if administration of fire is difficult or impossible. For example, fuel loads may be extremely high, the location may present difficulties (e.g., near a highway where smoke would present a hazard to motorists), or labor to administer the burn may be unavailable. Increasingly, people live in or near managed forests (Cohen 2000; Egan and Luloff 2000), and because of complaints about smoke and concerns about potential damage, managers increasingly are reluctant to burn or are having more difficulties obtaining burning permits.

Prescribed burnings that escape control are of special concern to landowners. For example, the May 2000 "Cerro Grande" fire that destroyed a large number of houses in Los Alamos and White Rock, New Mexico, began as a prescribed burn on the north rim of the Grand Canyon. Private landowners often have been the target of litigation related to unintended consequences of prescribed burning, and since the passage of the federal Tort Claims Act, even federal agencies are not immune from litigation over such matters. In contrast, drift of herbicides can be minimized by pre-planning applications using recently developed modeling tools such as AgDRIFT® (Teske 2000).

Regulations and guidelines at the local, state, and federal levels also may constrain a manager's ability to use fire. Many states have stringent requirements regarding weather conditions under which prescribed burning can and cannot be used. For instance, regulations in Texas prohibit the use of fire under conditions when smoke will present a hazard on any "pubic road, landing strip, or navigable water" or when it will affect a "sensitive receptor" (e.g., a residence, business, farm building, or greenhouse) (Texas Natural Resource Conservation Commission 2000). At the federal level, EPA's interim air quality policy on wildland and prescribed fire (Environmental Protection Agency 1998) also constrains the use of fire in order to regulate emissions of particulate matter and visibility impairments in the 156 mandatory Class 1 federal areas ("Areas of Great Scenic Importance"). Regional haze regulations that eventually will be promulgaed by EPA may further complicate prescribed burning. When air quality is an administrative concern, EPA's Interim Air Quality Policy on Wildland and Prescribed Burnings (Environmental Protection Agency 1998) explicitly states that "chemical treatments may be appropriate tools."

Economics

Costs obviously are an important consideration when selecting a habitat managment tool. Generally, prescribed burning costs less to apply per unit area than do herbicides. Average costs in the South during 1998 were $40.97/ha for prescribed burning and $178.70/ha for herbicide applications (Dubois et al. 1999). However, several other factors also should be considered when evaluating the cost of fire and herbicides. Multiple applications of prescribed burning over years or decades sometimes are required to achieve the same level of vegetation control that can achieved with one application of herbicides (Lautenschlager et al. 1998). Although liability costs and loss of growth do not occur every time a forest is burned, they could significantly affect the cost of prescribed burning in some situations and were not incorporated into estimates by Dubois et al. (1999). Even without considering these factors, the cost of applying fire has increased dramatically relative to the cost of applying herbicides. A cost index calculated by Dubois et al. (1991) for prescribed burning increased at an average annual rate of 10 percent between 1952 and 1988, over twice the rate for herbicide applications (Dubois et al. 1999).

Operability at Desired Spatial and Temporal Scales

In deciding whether to use fire or herbicides, managers also should consider factors related to time and space. For example, herbicides can be applied to individual plants, patches of vegetation within stands, and at the stand scale or larger. In contrast, fire is most easily applied at the stand or community levels. Herbicides sometimes immediately produce desired responses in plant communities (e.g., reduction of non-native species), while multiple applications of fire over several years may be required.

Conclusion

The choice of whether to use prescribed burning or herbicides for achieving wildlife management objectives depends upon many factors. For achieving some habitat objectives, herbicides probably are a preferred or partial alternative to fire. In other cases, fire is the most appropriate tool. However, prescribed burning sometimes cannot or will not be used because of concern about liability, smoke management difficulties, availability of labor, limited burning days, or other reasons. In such cases, herbicides may be the only tool available and must be used if biologists are to even partially address a wildlife-related objective. Generally, herbicides are most useful from a wildlife management prespective for shaping individual habitat components in well-defined areas. However, no habitat management tool, whether prescribed burning or herbicides, is best or even capable of addressing every wildlife management objective. Thus, we urge managers to retain access to an assortment of tools, including herbicides, and to use them in an integrated fashion.

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Fire and Bats in the Southeast and Mid-Atlantic: More Questions Than Answers?

Timothy C. Carter[1]*, W. Mark Ford*[2] *and Michael A. Menzel*[3]

Abstract.—The role and impact of fire in Southeastern ecosystems has changed dramatically from pre-European and early settlement times through present day. Regionally, pre-settlement fires were caused either by Native Americans throughout the year or by lighting-caused wildfires during the growing season. Today, much of the prescribed burning for forest and game management purposes occurs during the dormant or winter season in the South, whereas many ecological restoration or maintenance fires are conducted in the late dormant season through the growing season depending on the region and habitat type. Many bat species in the Southeast presumably have evolved in fire-dominated ecosystems with roosting strategies that limit their vulnerability to fire. Moreover, fire in any season that causes overstory tree mortality and creates snags suitable as bat roosts probably provide far more benefit to bats than do the negative impacts from burning. Dormant season burning may render tree/foliage-roosting bats vulnerable to fire in areas of the deep and mid-South where winter temperatures force prolonged periods of inactivity and roosts may be limited in highly fragmented or intensively managed forests. All bats that tree roost can be impacted by growing season burning if non-volant young are present. In the winter, cave-dwelling myotids seem the least vulnerable to negative impacts from wildfire and prescribed burning, although implications from the dramatic increase in late dormant-early growing season prescribed burning in the Appalachians and Interior Highlands to bats are unknown.

Introduction

Little or no research has been conducted on wildfire and prescribed burning impacts to bats and bat populations. Consequently, wildlife managers only can approximate fire effects on these taxa by examining fire impacts to habitat components important to bats. Although fire effects on wildlife and the ecological role of fire in certain Southeastern plant communities such as longleaf pine (*Pinus palustris*) forests are fairly well known (Landers et al. 1989; Landers et al. 1995; Brennan et al. 1998), only recently have wildlife managers started to understand even the most basic aspects of bat ecology and habitat use in the Southeast. For most bats, this understanding is minimal at best (Wilkins 1987; Watkins 1972; Shump and Shump 1982a; Shump and Shump 1982b; Kunz and Martin R. A. 1982; Kunz 1982; Thomson 1982; Webster et al. 1980; Jones 1977; Fenton 1997; Jones and Manning 1989). Bats are the second most diverse group of mammals in the Southeast and Mid-Atlantic with 18 species occurring north of "tropical" midsection of Florida (Whitaker and Hamilton 1998). To further complicate any assessment of the fire impact on bats, life histories for these 18 different species differ dramatically and each species potentially could be affected in profoundly different ways.

Prescribed burning benefits to bats in the Southeast and Mid-Atlantic are attributed to forest habitat modifications that alter or increase amounts and quality of roosting habitat, modify or improve foraging habitat and increase arthropod prey abundance. Within heavily wooded areas, fires that cause overstory mortality and create canopy gaps may allow bat species such as eastern red bats (*Lasiurus borealis*), big brown bats (*Eptesicus fuscus*), or other edge-open ecotone specialist bats to forage more effectively (Edwards et al. 2000). Also, depending on fire intensity, tree species susceptibility, and tree canopy position, potential roost snags or roosting cavities for big brown bats, northern long-eared bats (*Myotis septentrionalis*), Indiana bats (*Myotis sodalis*) and evening bats (*Nycticeius humeralis*) can be created. However, newly created snags may not be immediately usable by bats. Burning is not an immediate panacea to a paucity of bat roosting substrate because cavity roosts or exfoliating bark roosts can take considerable time to appear and may not be available for one or more seasons depending on forest community composition and regional climate.

Burn Season

Most plausible impacts of growing season burns for bats are poorly documented. Because there are more species of bats using the forest and because parturition occurs for most eastern bats at this time, total mortality could be greater than for dormant season burns. Snags serving as bat roosts could be consumed by burning and during intense fires, roosting bats could be killed. Even without direct mortality, large-scale habitat alteration or habitat destruction could be detrimental to resident bats during a time when females presumably are stressed by the physiological demands of parturition and lactation. The life histories of many bat species have several attributes that could be considered adaptive to growing season fires. During summer months, bats are able to arouse quickly as the difference between the ambient

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[1] Department of Zoology, Southern Illinois University, Carbondale, IL 62901

[2] Northeastern Research Station, USDA Forest Service, Parsons, WV 26287

[3] Division of Forestry, West Virginia University, Morgantown, WV 26506

temperature and active body temperature of bats is less. Most bats are quick and highly vagile, flying at speeds > 30 kmph (Patterson and Hardin 1969) so that escape and relocation to unburned areas easily can occur. Most bat species utilizing trees and snags have multiple roosts throughout the forest (Sasse and Pekins 1996; Callahan et al. 1997; Menzel et al. 1998; Foster and Kurta 1999, Menzel et al. 2001), providing alternate roosts should the current roost be destroyed by fire. Lastly, most eastern bat species are able to carry their young for some time after they are born (Davis 1970). Conceivably, this would allow females to relocate their young if their nursery roost became endangered by fire.

Although not necessarily true for the deep South, in the mid-Atlantic and mid-South, dormant season burns occur when there are far fewer bats in the forest than in the growing season. Most dormant season burns tend to be controlled prescribed fires with a relative low intensity as currently practiced for pine plantation management or bobwhite quail (*Colinus virginianus*) management. Overstory mortality generally is low and snag creation often is limited. These lower intensity fires also may consume fewer existing snags than would more intense fires at other times in the year. Nonetheless, bats using forests burned during winter months may not be fully adapted to this type of rather "unnatural" disturbance. During winter months, many eastern bats hibernate or enter prolonged periods of inactivity during cold snaps. Bats may take up to 30 minutes to arouse from hibernation (Thomas et al. 1990), making it difficult for those not hibernating in more protected cave or mine refugia to reach a body temperature and activity level sufficient to escape a fire threatening a roost. Dormant season burning is most prevalent in the deep South and the Coastal Plain areas of the mid-Atlantic where winter temperatures generally are warm enough to ensure year-round presence of bats. In upland hardwood forests in the mid-Atlantic states, such as West Virginia and Virginia, burning for ecological restoration efforts to promote oak (*Quercus* spp.) regeneration (Brose et al. 1999) takes place from mid-April to mid-May. During this time, many bat species are finished hibernating and already could be using these upland forests for roosting and foraging.

Solitary Foliage-roosting Guild

Bat species in this guild use the forest exclusively for roosting. Regionlly, this includes the eastern red bat, Seminole bats (*Lasiurus seminolus*), and hoary bat (*Lasiurus cinereus*) that roost in tree foliage throughout the year. Eastern pipistrelles (*Pipistrellus subflavus*) roost within tree foliage during the summer (Shump and Shump 1982; Menzel et al. 1998, Veilleux 1999). The northern yellow bat (*Lasiurus intermedius*) generally roosts in Spanish moss (*Tillandsia usneoides*; Webster et al. 1980; Menzel et al. 1999). With the exception of females nursing young, all individuals in this guild roost solitarily. We hypothesize that these roosting habits translate into few or minimal fire effects on this group as disturbance events probably affect fewer animals at any single time as opposed to colonial-roosting bats. Furthermore, their habit of foliage roosting permits quick perception of surroundings and allows for faster response times. In largely forested landscapes, there are almost infinite amounts of available roosts for alternate use. However, in highly fragmented landscapes, fire and smoke may drive these bats to alternate forest patches as witnessed by Rodrigue et al. (2001), increasing their vulnerability to daytime predation as bats cross non-forested areas.

During winter months, this roosting guild is composed exclusively of Lasiurines, as the eastern pipistrelle retreats to cave or mine hibernacula. Of bats that continue forest roosting throughout winter such as the eastern red bat, some individuals migrate to the deep South and continue similar roosting and activity habits, whereas others individuals staying in the mid-Atlantic and upper South will periodically retreat to the leaf litter on the forest floor to escape cold snaps. Eastern red bats have been observed flying up from the leaf litter in front of dormant season fires (Saugey et al. 1998; Moorman et al. 1998). Whether or not eastern red bats or any other foliage roosting bats experience significant annual mortality from dormant season burning is unknown. An extensive study of the winter roosting habits and the possible effects of fire on this roosting guild has not been conducted.

Bark and Cavity Roosting Guild

This guild uses tree cavities and exfoliating bark for roosts. Generally, these bat species form small roost colonies that render them more susceptible to fire effects than solitary foliage roosting bats. As a result, even a single, localized fire event potentially could impact more individual bats. Cavity roosting bats include the big brown bat, evening bat, Rafinesque's big-eared bat (*Corynorhinus rafinesquii*), northern long-eared bat, and silver-haired bat (*Lasionycteris noctivagans*). Regionally, evening bats often are found roosting in upland pine forests (Menzel et al., 2001) where both natural and anthropogenic fires are common. Because these bat species roost internally within cavities, their perception and response time necessary for exit and escape may be delayed. Bark roosting species include the endangered Indiana bat, little brown bat (*Myotis lucifugus*), and at times, the northern long-eared bat (Menzel et al. in press). In the northern parts of its distribution, the southeastern bat (*Myotis austroriparius*) will use exfoliating bark and hollow trees, whereas in the southern portion it also utilizes caves, bridges and other structures as roosts (Gardner et al. 1992). The small to medium sized maternity colonies formed under exfoliating bark in trees and snags by this roosting guild are vulnerable to fire disturbance in several ways. Snag roosts, depending on age and condition and fire condition can be highly combustible. However, most of these species use multiple summer roosts that combined with their ability to carry young in flight might offset this significant liability (Davis 1970). Also, many of these bats often roost in riparian areas where prescribed burning generally is not applicable and where wildfire risks are slight (Gardner et al. 1991; Gardner et al. 1992; Foster and Kurta 1999).

During winter, all the myotids as well as big brown bats and eastern pipistrelles in the foliage roosting guild move to hibernacula in caves, mines, and large rockhouses and clifflines with cave-like characteristics. Silver-haired bats winter from the lower Ohio Valley and mid-Atlantic south into the mid-South where they switch to rock crevice and solitary bark roosting habits. Evening bats migrate to the deep South and coastal regions and continue using tree cavities for roosts. We hypothesize that fire effects on evening bats and silver-haired bats during winter months would be similar to those of the summer months with the added caveat that these animals could be in a prolonged torpor with slower reaction times.

Cavernous Roosting Guild

Of the bats that use caves and mines, few use underground roosts during the summer months. Most use forests (cavities and exfoliating bark) during the summer months. Bats that summer-roost in caves or mines include southeastern bats, gray bats (*Myotis grisescens*) and Townsend's big-eared bats (*Corynorhinus townsendii*). Bachelor colonies of northern long-eared bats and Indiana bats occasionally have been found in caves and mines during the growing season. Although subterranean roosts offer direct protection from flames, fire near occupied underground roosts can impact bats both directly and indirectly. Fire that alters surrounding vegetation such that cave or mine airflow is modified potentially could affect entrance and chamber microclimate (Richter et al. 1993). Additionally, habitat changes around caves or mines may impact forgaging bats during pre-hibernation swarming or post-hibernation pre-dispersal, two feeding periods when bats must gain critical fat reserves. Smoke impacts to cavernous roosting bats depend on a cave's airflow characteristics. Depending on the season and air temperature, cave and mine entrances serve as entrances and exits for warm air or cold air (Tuttle and Stevenson 1977). A fire that creates smoke up wind from a "breathing" entrance potentially could fill the cave with smoke. In the late spring and summer, maternity colonies of gray bats or Townsend's big-eared bats that seek warm areas in a cave could be directly influenced by smoke drawn into the portions of a cave serving as a warm air trap or reservoir. During the winter months, the cavernous roosting guild also expands to include the rest of the myotids, eastern pipistrelles, big brown bats and Rafinesque's big-eared bats. Most caves and mines used as hibernacula tend to trap colder air in the winter, so smoke entry into the system probably would be less likely to occur. Nonetheless, if smoke from a nearby fire is drawn into the hibernaculum, the potential for a catastrophic event affecting large groups of bats is a possibility. Mortality from smoke inhalation or decreased fitness from premature overwinter disturbance and arousal are real possibilities. The increased interest in the use of fire to manage upland hardwood forests in the mid-South and mid-Atlantic where karst topography and cave formation occurs merits expanded fire-ecology research.

Crevice Roosting Guild

Regionally, the myotids, Rafinesque's big-eared bat, and Townsend's big-eared bat occasionally will utilize rock outcrop and clifflines during the growing season for roosting habitat. Also, silver-haired bats have been observed using rock outcrops as winter roosts in Arkansas (D. Saugey, USDA Forest Service, pers. comm.). However, during the summer months, the small-footed bat (*Myotis leibii*) almost exclusively uses cracks and chimneys in large rock outcrops and clifflines (Whitaker and Hamilton 1998). Depending on localized geology, these roosts can be found in a variety of forested habitats and conditions in the region. Within the mountainous portions of the Appalachians in the mid-South and mid-Atlantic, rock outcrops occurring along xeric ridgelines often are covered by dry-site oak communities or fire-adapted pitch pine (*Pinus rigida*)-table mountain pine (*Pinus pungens*) overstories with extremely flammable ericaceous shrub layers. Although regional interest in prescribed burning in these systems to maintain these relictual pine types is increasing (Vose et al. 1997; Welch and Waldrop 2001), the impacts to bats such as the small-footed bat are unknown.

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Wildlife, Habitat, and Prescribed Fire in the Southeastern Coastal Plain: Past, Present, and Future

Leonard A. Brennan[1]

Abstract.—Fire has been an imperative ecological force that has shaped the natural history and culture of the Southeastern Coastal Plain. In the past, humans have influenced the " natural" role of fire in the southeast, first by Native Americans, then by Spanish and other European settlement, and finally by modern culture. As human culture developed, fire, and how it influenced wildlife populations on both the landscape and local scales, became more and more removed from the set of environmental conditions that originally allowed it to shaped the biota of the southeastern landscape. Presently, broad scale habitat fragmentation from urbanization, agriculture, and intensive forestry, has eliminated fire from much of the southeast. Also, at the present time, virtually all of the vertebrates in the southeast that are declining, threatened, or endangered require open forest conditions maintained by frequent fire. In the future, most fire in the southeast will probably be relegated to isolated fragments of public lands, ecological preserves, and private hunting plantations. However, there are some developments that have emerged from research, as well as management experience and common sense, that have promise. For example, fire ecologists and managers have learned to pay respect to the use of fire is emerging. Such an approach transcends traditional single species, game management objectives, and incorporated consideration of how keystone native plants, arthropods, and other nongame vertebrates, respond to variation in seasonal fire applications, as well as to variation in fire return intervals.

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[1] Tall Timbers Research Station, Tallahassee, FL 32312

The Role of Fire in Riparian Zones of the Northern Rocky Mountains

Elaine K. Sutherland[1] *and Kevin McKelvey*[1]

Abstract.—While the importance of riparian systems in the northern Rocky Mountains as sources of productivity and diversity is recognized, there is little information about the interaction between pattern and process. To sustain these areas, we need to understand the characteristics of disturbance processes and how they result in patterns in these systems. There is mounting evidence that simply protecting riparian areas from fire and other disturbances results in unsustainable riparian conditions and deterioration of habitat for wildlife. Recent high-intensity fires, which in the past were probably uncharacteristic of the upland vegetation surrounding riparian zones, can result in soil erosion and sediment loading that can damage aquatic systems. However, low-intensity fires play an important role in determining composition and structure in upland and streamside vegetation, and consequently in contribution of vegetation debris to the aquatic system. Without low-intensity fire, uplands and streamsides succeed to shade-tolerant coniferous species, with consequently reduced dominance or loss of early-successional deciduous trees and shrubs. These altered conditions can have important consequences for habitats of terrestrial and aquatic fauna.

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[1] Rocky Mountain Research Station, USDA Forest Service, Missoula, MT 59807

The Restoration of Oak-Hickory Forests in the Central Hardwoods: Results of a Landscape-scale Prescribed Burning Experiment in Ohio

Elaine K. Sutherland[1] *and Todd F. Hutchinson*[2]

Abstract.—Oaks have been an important component of eastern deciduous forest communities since the end of the last glaciation, 10,000 years BP. There is wide recognition that oaks are highly fire adapted, and that fire played an important role in the ecology of oak forests in the past, particularly in promoting the dominance of oak in regeneration layers. In southern Ohio, forest composition and structure has changed since aggressive fire protection was implemented. Although oaks still dominate the overstory, advanced regeneration is dominated by species such as yellow-poplar, maples and blackgum. The objectives of this project are to determine ecological response of mixed-oak communities in southern Ohio to prescribed underburning under frequent and infrequent fire regimes. Four study areas, ~ 90 ha each, were selected in 1994 in the hilly country of the unglaciated Allegheny Plateau. These areas are dominated by oak forests > 80 years old. Study areas were stratified by an estimator of available soil moisture based on landscape physiographic characteristics and soil water holding capacity. This integrated moisture index (IMI) is a good predictor of vegetation composition, as well as songbird species occurrence. Study areas and plots were installed in 1994 and pre-burning data were gathered in 1995. The four study areas are divided in thirds; two early spring burning treatments and an unburned reference. The frequent treatments were burned annually (1996-1999) and infrequent treatments were burned twice (1996 and 1999). Physical and biological attributes of the forests were fully monitored between 1995 and 1999. These low-intensity fires had significant effects on soil and litter distribution and composition and structure of herbaceous and small diameter vegetation, but many of these effects likely will be short-lived. There were consequent changes in ground-nesting bird populations, but not in those nesting elsewhere. Insect communities were little altered.

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[1] Rocky Mountain Research Station, USDA Forest Service, Missoula, MT 59807

[2] Northeastern Research Station, USDA Forest Service, Delaware, OH 43015

Fire Ecology of Marshes and Canebrakes in the Southeastern United States

Cecil Frost[1]

Abstract.—Most marshes in the United States can be considered fire communities. With the exception of true salt marsh (salinity 3-4%), maintenance of species diversity in marshes is dependent upon frequent fire. Plant species diversity in marshes decreases as salinity increases and increases as fire frequency increases. Canebrake, another fire community, once covered at least 10 million acres in stream bottomlands and peatlands of the southeastern United States. This comprised critical habitat for a number of rare plant and animal species. Far less than 1% of this habitat remains, canebrakes having disappeared after fire was removed from the landscape. When fire is kept out, canebrake succeeds to multistoried wooded communities such as bottomland hardwoods, pocosin, pond pine forest, red maple forest, and bay forest. Rare, fire-dependent plants native to canebrakes include golden sedge (*Carex lutea*), *Lilium iridollae* and *Lilium gazarubrum*. Important wildlife species include Swainson's warbler, Bachman's warbler, and a butterfly, St. Francis satyr. Canebrake also is a fire frequency indicator community. A historical record of canebrake for a particular location indicates that that region once experienced a frequent fire regime. Studies of succession in canebrakes indicate that the original fire frequency ranged mostly between 2-8 years between fires. Protection and restoration of canebrake requires management with fire, and has been almost entirely overlooked in site management and conservation planning.

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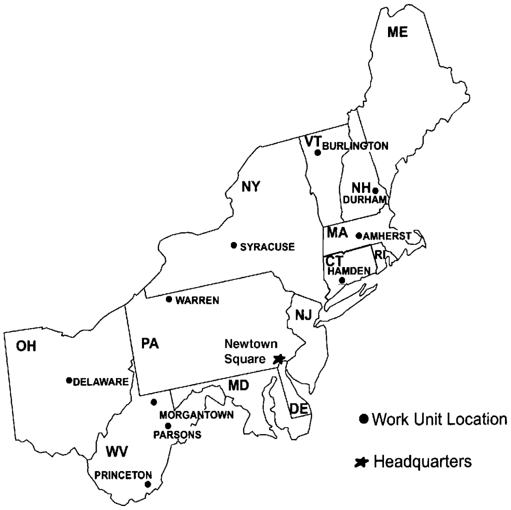
[1] North Carolina Plant Conservation Program, Department of Agricultural and Consumer Services, Raleigh, NC 27611

Ford, W. Mark; Russell, Kevin R.; Moorman, Christopher E., eds. 2002. **Proceedings: the role of fire for nongame wildlife management and community restoration: traditional uses and new directions**; 2000 September 15; Nashville, TN. Gen. Tech. Rep. NE-288. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station. 145 p.

Contains 16 papers and 4 abstracts on the use of fire for nongame wildlife management and rare plant community management in restoration in the United States with an emphasis on forested systems.

Keywords: alternative management, community restoration, history of fire, prescribed fire, nongame wildlife



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