POSTFIRE CHARACTERISTICS OF EXISTING AND POTENTIAL BLACK-CAPPED VIREO (*VIREO ATRICAPILLUS*) NESTING HABITAT ON THE WICHITA MOUNTAINS WILDLIFE REFUGE, OKLAHOMA

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PREFACE

Each chapter of this thesis is complete for publication in a specific scientific journal. Chapter I is written in the style of Conservation Biology, and Chapter II is in the style of The Southwestern Naturalist. Additional data and study site maps are included in a 1995 report entitled "Effects of fire on existing and potential nesting habitat of the black-capped vireo (*Vireo atricapillus*) on the Wichita Mountains Wildlife Refuge". This report can be obtained from either the United States Fish and Wildlife Service, Wichita Mountains Wildlife Refuge, Oklahoma, or the Oklahoma Cooperative Fish and Wildlife Research Unit at Oklahoma State University.

I greatly appreciate the guidance of Drs. James H. Shaw, Terrence G. Bidwell, and David M. Leslie, Jr. during my research. The outstanding scholarship of Jim Shaw was an inspiring example for me. I am also grateful to Joe Kimball of the Wichita Mountains Wildlife Refuge for his assistance throughout this project. I thank my parents, Norman and Marion Greenman, for their extreme patience and support during my long college career. Finally, I thank my friend Marcus Koenen for helping in countless ways.

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CHAPTER I

HABITAT AVAILABILITY FOR AN ISOLATED POPULATION OF THE ENDANGERED BLACK-CAPPED VIREO

ABSTRACT: Nesting habitat fragmentation has isolated populations of the endangered black-capped vireo (*Vireo atricapillus*). It is uncertain whether suitable habitat remains for several protected populations that are now increasing. We described structural characteristics of black-capped vireo nesting habitat on the Wichita Mountains Wildlife Refuge, Oklahoma, and compared existing and potential habitat to determine habitat availability for a growing, isolated vireo population. We found statistically significant differences among some treatments for canopy height, standard deviation of canopy height, and horizontal heterogeneity variables, but none of the habitat characteristics indicated a difference between vireo-occupied and vireo-unoccupied treatments. This suggested that the extant population was not occupying all available habitat and the Refuge contained a sufficient amount of habitat to accommodate a population larger than the current one. Colonization of unoccupied habitat may have been slowed by small population size and black-capped vireo behavior. Some unoccupied habitat may have been less suitable for nesting because lack of soil moisture and nutrients limited woody vegetation growth. The unburned treatment may have been less suitable because it contained high frequencies of eastern redcedar (*Juniperus virginiana*) rather than blackjack oak (*Quercus*

marilandica), which is the preferred nest tree species. The Refuge black-capped vireo population may remain isolated from other populations due to the difficulty of re-establishing a network of occupied habitat patches that includes the Wichita Mountains. Habitat patches outside the Refuge are too widely scattered to have colonization sources, and vireos dispersing from the Refuge have a low probability of encountering remaining fragments of habitat.

INTRODUCTION

Habitat loss is a major cause of extinction (Thomas 1994) and is the primary cause of endangerment for 65% of endangered bird taxa (King 1977). Habitat loss often occurs in a fragmentary pattern that reduces available habitat, decreases interchange among populations, and leaves remaining isolated populations vulnerable to extinction (Pickett & Thompson 1978; Terborgh & Winter 1980; Wilcove et al. 1986; Robinson 1992; Angermeier 1995). These effects may be especially severe for species that have patchy distributions (Wilcox 1980; Wiens 1985; Wilcove et al. 1986), such as the endangered black-capped vireo (*Vireo atricapillus*). Prior to Euro-American impacts, its nesting habitat probably occurred as patches throughout the landscape. Vireo populations subsequently declined or became extinct as patches disappeared. Currently, several populations are increasing with intensive management (Grzybowski 1993; J. Cornelius, personal communication). For this endangered species and others that are experiencing some recovery success, a crucial conservation question is whether sufficient habitat remains to accommodate their growing populations.

The black-capped vireo is a neotropical migratory passerine. In 1987, the United States Fish and Wildlife Service listed the species as endangered because of reductions in both breeding range and abundance. Population declines are due to nesting habitat loss and nest parasitism by the brown-headed cowbird (*Molothrus ater*) (Ratzlaff 1987). The historic breeding range extended from Kansas through Oklahoma and Texas to Coahuila, Mexico (Graber 1957; U.S. Fish & Wildlife Service 1991). This limited range occurs in the transition between eastern deciduous forest and the southern plains, and south to mountain slopes in the arid Chihuahuan zone. Black-capped vireos are now extirpated or declining in more than 50% of their former range. Only two populations remain in Oklahoma, with most of the birds concentrated in a 10-km^2 area of the Wichita Mountains Wildlife Refuge (Grzybowski 1993). As a result of habitat protection and cowbird management, the Wichita Mountains population is increasing, but it is isolated from other remaining populations and thus is vulnerable to extinction (Barlow 1980; Grzybowski 1991; Maurer & Heywood 1993).

Black-capped vireo nesting habitat contains scrubby, woody plants that have foliage within 3 m of the ground. This habitat is produced by arid climate, fire, and edaphic and topographic factors. Habitat has been destroyed directly by land-clearing and livestock browsing and indirectly by alteration of fire regimes (Graber 1957; U.S. Fish & Wildlife Service 1991). Breeding vireos are found most often on areas that have recently burned (Graber 1957; Benson & Benson 1990; Grzybowski 1993; J. Cornelius, personal communication). The Wichita Mountains Wildlife Refuge contains areas of unoccupied habitat that structurally and floristically appear to be similar to areas occupied by black-capped vireos. Our objectives were to describe structural characteristics of black-capped vireo nesting habitat on the Refuge and compare existing and potential habitat to determine habitat availability for this growing population. To accomplish this, we tested the null hypothesis that there are no differences in the characteristics of vireo-occupied and vireo-unoccupied treatments.

STUDY AREA

The 24,000-ha Wichita Mountains Wildlife Refuge contains Precambrian granite and gabbro remains of an ancient mountain range. Elevations range from 347 to 756 m. Average annual precipitation is 808 mm, but it varies greatly, and drought periods are common (Wichita Mountains Wildlife Refuge weather records). The Refuge encompasses a patchy mosaic of mixed-grass prairies, scrub oaks (*Quercus marilandica and Q. stellata*), hardwood forests, rocky escarpments, and talus slopes (Buck 1964; Crockett 1964). Large herbivores include bison (*Bison bison*), elk (*Cervus elaphus*), white-tailed deer (*Odocoileus virginianus*), and Texas longhorn cattle (*Bos taurus*). Fires may occur in any season; prescribed fires are conducted by Refuge personnel, and wildfires are ignited by visitors and activity on the adjacent Fort Sill military reservation.

METHODS

Sampling Techniques

We installed 106 permanent line transects (Cain & Castro 1959) at randomly selected points in an unburned treatment and treatments burned 1, 2, 5, and 15 years prior to the study. The 5-year-old burn treatment was divided into north and south-facing slope aspects. The 15-year-old burn treatment was divided into existing and potential nesting habitat. The 5-year-old burn treatments also were existing habitat (Table 1). Existing habitat was defined as areas where vireo breeding territories occurred yearly since study of the population began in 1985. Potential habitat was defined as areas that appeared similar to existing habitat but did not contain territories. Transects were X-shaped and consisted of a central steel post stamped at the top with an identification number, with steel stakes at each of the 4 endpoints. Transect axes were oriented along the cardinal directions, and each transect had a total length of 100 m. We sampled 100 consecutive points at 1-m intervals on the line, proceeding from south to north for points 1 to 50 and from east to west for points 51 to 100.

Transects were sampled for two consecutive years. From mid-June to early August, 1993 and late June to late July, 1994, we collected data at each point by recording contacts with an extendable 7.5-m graduated range pole placed vertically on the ground, as modified from Mills et al. (1991). Ground cover was recorded at the pole base; one canopy cover contact was recorded in each dm-interval within 2 m of the ground, as well as in each subsequent 0.5-m height interval. Cover above 7.5 m was recorded as present if an imaginary extension of the pole contacted vegetation. When more than one contact occurred in an interval, the highest contact was recorded. Abiotic components were categorized as bare ground, gravel, rock, dead wood, and grass, sedge, forb, or tree litter (Rotenberry & Wiens 1980). Vascular plants were recorded by scientific name (U.S.D.A. Soil Conservation Service 1982); herbaceous plants that were unidentifiable because of phenological stage were recorded as unidentifiable grass, sedge, or forb. Microphytes were categorized as lichen, moss (Bryophyta and Lycophyta:*Selaginella* sp.), and cyanobacteria.

Variable Calculations

Because plant species composition varies over the breeding range and structure is important in black-capped vireo habitat selection (Cody 1985; Grzybowski et al. 1994), our focus was mainly on structural habitat characteristics. For each treatment (Table 1), we calculated mean values for variables important for nesting black-capped vireos (Graber 1957; Grzybowski et al. 1994). These include height of both total canopy (all biotic and abiotic contacts) and woody species canopy cover; vertical density (Rotenberry & Wiens 1980) at nest levels (the first 2 m above the ground) of both total contacts and woody species contacts; and absolute frequency of woody species canopy cover (expressed as percent). We also calculated several heterogeneity indices. Vertical heterogeneity indices (indicating canopy height variation) for both total and woody species canopy were calculated as the means of transect standard deviations of canopy height. Two variations of a horizontal heterogeneity index, indicating patchiness or variation in the horizontal distribution of cover types, were modified from Grzybowski et al. (1994). These were calculated using the highest interval contact at each point along the line. The first was a count of the number of horizontal changes among 3 cover type categories (woody species, herbaceous species, and other, which includes abiotic components and microphytes). The second was a count of the number of horizontal changes between 2 cover type categories (woody species and the other 2 cover types combined).

Statistical Analysis

For each variable, differences among treatment means were determined by 1-way analysis of variance (ANOVA), using a significance level of $\alpha = 0.05$ (Freund & Wilson 1993). For variables having unequal treatment variances, as determined by Levene's test (Snedecor & Cochran 1980), ANOVA was performed on rank-transformed data (Conover & Iman 1981). Post hoc comparisons were made by Tukey's procedure (Freund & Wilson 1993), using Kramer's approximation to account for the unequal sample sizes (SAS Institute, Inc. 1985). We used SAS computer software to perform the analyses (SAS Institute, Inc. 1985).

RESULTS AND DISCUSSION

We noted significant differences among some treatments for canopy height, standard deviation of canopy height, and horizontal heterogeneity variables

(Tables 2 and 3). However, because of the variation and patchiness within treatments, none of the variables indicated a difference between vireo-occupied and vireo-unoccupied treatments. Most importantly, this suggests that the extant vireo population was not occupying all available habitat and that the Refuge contained a sufficient amount of habitat to accommodate a population larger than the current one. Although the occupied and unoccupied treatments were not statistically different, nonetheless there were differences between individual treatments that may affect habitat suitability for black-capped vireos.

Existing habitat

Black-capped vireo habitat on the Wichita Mountains Wildlife Refuge occurred on hill and mountain slopes and consisted of woody vegetation patches within a matrix of grass and rock. Woody vegetation with an average height of 2-3 m made up 33-48% of the cover. Blackjack oak was the dominant woody species within 2 m of the ground; above 2 m, taller post oaks added to canopy structural heterogeneity in the vertical dimension (Greenman et al. 1995). The prolific sprouting of blackjack oak in response to fire produced woody vegetation vertical density of 61-91 contacts/transect in the 0-1 m height interval and 48-63 contacts/transect in the 1-2 m height interval.

Differences among vireo-occupied treatments indicated that the species tolerated variation in habitat. Woody vegetation was taller, more vertically heterogeneous, and in larger patches in the north-facing 5-year-old burn than in the south-facing 5-year-old burn and especially the vireo-occupied 15-year-old burn. The south-facing aspect of the 5-year-old burn had higher frequencies of rock cover (40-43%) than the north-facing aspect (23-25%) (Greenman et al. 1995) and denser nest-level woody vegetation than the north-facing aspect. Habitat was more open and woody patches more widely spaced in the vireo-occupied 15-year-old burn than in the 5-year-old burn treatments.

Potential Habitat

Variables in the vireo-unoccupied 1-year-old burn were more similar to occupied treatments than other unoccupied treatments; therefore, it probably contained suitable habitat. Vireo expansion from the occupied habitat to this area would help alleviate vulnerability of this population to stochastic occurrences such as drought, fire, and detrimental genetic or demographic fluctuations. Although this treatment was less than 2 km from existing habitat, apparently the 1993-94 population was not at a critical size necessary for colonization of unoccupied areas. Dispersing vireos are not likely to find mates away from the core habitat (Pease & Gingerich 1989), which is compounded by the typically clumped distribution of black-capped vireo territories and the greater reproductive success in large groupings than in small ones (Graber 1957; U.S. Fish & Wildlife Service 1991). These behavioral factors may slow colonization of unoccupied habitat.

Habitat in the vireo-unoccupied 15-year-old burn, and to a lesser degree the vireo-unoccupied 2-year-old burn, may have been less suitable than habitat in occupied treatments because of differences in vegetation structure. High horizontal heterogeneity in conjunction with the low frequency of woody species canopy cover indicated an open habitat with widely spaced woody patches. Both total and woody species canopy had short height, low vertical density in the 1-2 m height interval, and low vertical heterogeneity, indicating a shorter, more homogeneous vertical dimension than in vireo-occupied treatments. The unoccupied portion of the 15-year-old burn had not developed habitat comparable to the occupied portion but instead was similar to the 2-year-old burn adjacent to it. This suggests that most growth was confined to the first several postfire growing seasons. This area may have had low levels of soil moisture and nutrients because of shallow soil, which severely restricted woody vegetation growth and prevented development of suitable habitat.

Habitat in the vireo-unoccupied unburned treatment may have been less suitable due to both vegetation structure and plant species composition. Low horizontal heterogeneity in conjunction with low frequency of woody species canopy cover indicated habitat more open and with fewer woody patches than in vireo-occupied treatments. The woody species canopy also was taller than vireo-occupied treatments, and tall trees often did not have the nest-level foliage found in scrub trees. The small sample size produced inconsistent vertical density data between years, so it is uncertain whether nest-level densities were low. However, woody densities appeared low in the field.

Unlike vireo-occupied treatments, the unburned treatment was dominated by eastern redcedar *(Juniperus virginiana)* rather than blackjack oak (Greenman et al. 1995). Blackjack oak was the primary nest tree species on the Refuge, and black-capped vireos select it over eastern redcedar for nest sites (U.S. Fish and Wildlife Service 1991). Eastern redcedar is an aggressive invader in the absence of fire (Arend 1950; Blewett 1986; Stritzke & Bidwell 1990). Habitat in the unburned treatment may have been less suitable due its plant species composition at nesting heights.

Conservation Implications

Some endangered species have no remaining habitat to recolonize, but others do, and determining habitat status would enhance their conservation. For example, it is unclear why two other fire-associated endangered bird species, the Florida scrub jay (*Aphelocoma coerulescens coerulescens*) and the Kirtland's warbler (*Dendroica kirtlandii*), are absent from apparently suitable habitat (Woolfenden & Fitzpatrick 1984; Mayfield 1993). In the case of the black-capped vireo, we found that there was surplus suitable habitat available for the growing population on the Wichita Mountains Wildlife Refuge. This offers hope for attaining an Oklahoma population of at least 500 pairs, one of the goals of the black-capped vireo recovery plan (U.S. Fish & Wildlife Service 1991).

Regardless of the size of the Wichita Mountains population, two obstacles continue to isolate it from other black-capped vireo populations. First, remaining empty habitat patches in the Oklahoma breeding range may be too distant from colonization sources (see Pickett & Thompson 1978; Wilcox 1980), which makes it difficult to re-establish a network of occupied habitat patches that includes the Wichita Mountains. Second, because black-capped vireo habitat patches often shift spatially and temporally, the United States Fish and Wildlife Service has not designated critical habitat for the species (Ratzlaff 1987). Thus, there has been no protection or restoration of potential habitat outside the Refuge. The result is that habitat remains fragmented, and vireos dispersing from the Refuge have a low probability of encountering suitable patches and establishing new populations.

Black-capped vireo habitat has undergone fragmention (U.S. Fish & Wildlife Service 1991), but the process can be reversed. If the cooperation of private and State land managers can be obtained, fire can be used to restore a strategic distribution of habitat patches radiating from the Refuge. This would improve recovery of the species by linking the Wichita Mountains population with populations in Oklahoma and Texas. Although the Wichita Mountains habitat is not yet saturated, such habitat management may be necessary now. For unknown reasons, the percentage of juveniles returning from the wintering range has been extremely low (Grzybowski 1993). It is possible these birds are already dispersing from the Refuge and are essentially lost from the population (Pease & Gingerich 1989). Preventing this loss by providing more habitat might promote a more rapid black-capped vireo increase than presently is occurring in Oklahoma.

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Table 1. Summary of treatments studied during 1993 and 1994 on the Wichita Mountains Wildlife Refuge, Oklahoma, U.S.A.

Code	*No. Transects*	*Description*
V5N	20	Existing black-capped vireo habitat, north-facing aspect. Burned December, 1988, five years prior to study; 1,581 ha wildfire.
V5S	20	Existing black-capped vireo habitat, south-facing aspect. Burned December, 1988, five years prior to study; 1,581 ha wildfire.
V15	20	Existing black-capped vireo habitat. Burned March, 1978, fifteen years prior to study; >6,070 ha wildfire.
1	20	Potential black-capped vireo habitat. Burned February, 1992, one year prior to study; 1,027 ha prescribed fire.
2	13	Potential black-capped vireo habitat. Burned March, 1991, two years prior to study; 851 ha wildfire.
15	7	Potential black-capped vireo habitat. Burned March, 1978, fifteen years prior to study; >6,070 ha wildfire.
NOBURN	6	Potential black-capped vireo habitat. Unburned. Possibly burned March, 1959; 111 ha wildfire. Ten transects were sampled both study years; 4 were deleted from analyses after burning in April, 1994; 931 ha wildfire.

Table 2. Mean values for structural characteristics of total canopy in existing and potential black-capped vireo nesting habitat on the Wichita Mountains Wildlife Refuge, Oklahoma, U.S.A.[a]

			Vertical Density (No. Contacts)						*No. Cover Type*	
			Height Interval				*Std. Deviation*		*Changes Among*	
	Height (m)		*0-1 m*		*1-2 m*		*Of Height (m)*		*3 Categories*	
Treatment[b]	*1993*	*1994*	*1993*	*1994*	*1993*	*1994*	*1993*	*1994*	*1993*	*1994*
V5N	1.8b	1.7b	285	262	52	49	1.6b	1.6b	27b	26b
V5S	1.1bc	1.1bc	271	245	58	55	1.3ab	1.3ab	31abc	32ab
V15	0.9acd	0.8ac	252	213	63	51	1.0ac	1.0a	33abc	36ac
1	1.2abc	1.1abc	265	235	50	47	1.2abc	1.2ab	30ab	31ab
2	0.7acd	0.6ac	251	230	46	36	0.9ac	0.8a	36ac	40c
15	0.4d	0.4a	244	202	28	30	0.6c	0.7a	41c	40ac
NOBURN	1.0abcd	0.9abc	242	194	48	59	1.0abc	1.0ab	31abc	34abc

[a] *Means (within column) followed by different letters are significantly different (α = 0.05).*
[b] *V = existing black-capped vireo habitat; N and S = north and south-facing slope aspects; numbers = years since burned.*

Table 3. Mean values for structural characteristics of woody species canopy in existing and potential black-capped vireo nesting habitat on the Wichita Mountains Wildlife Refuge, Oklahoma, U.S.A.[a]

	Woody Canopy (%)		Height (m)		Vertical Density (No. contacts) Height interval 0-1 m		Vertical Density (No. contacts) Height interval 1-2 m		Std. Deviation of Height (m)		No. Cover Type Changes Between 2 Categories	
Treatment[b]	*1993*	*1994*	*1993*	*1994*	*1993*	*1994*	*1993*	*1994*	*1993*	*1994*	*1993*	*1994*
V5N	48	47	3.0b	2.9b	72	65	51	48	1.5b	1.5b	17	17
V5S	43	41	2.3bc	2.3bc	91	90	57	53	1.3ab	1.4ab	17	18
V15	35	33	1.9acd	2.0ac	72	61	63	50	1.1a	1.1ab	16	17
1	36	36	2.4abc	2.4abc	62	62	49	47	1.1a	1.2ab	16	16
2	32	32	1.6acd	1.5ac	86	84	46	35	0.9a	1.0a	18	20
15	22	22	1.2d	1.3a	74	65	27	30	0.8a	1.0ab	18	18
NOBURN	28	30	3.3abcd	3.4abc	67	63	47	59	1.0ab	1.1ab	9	10

[a] *Means (within column) followed by different letters are significantly different (α =0.05).*
[b] *V = existing black-capped vireo habitat; N and S = north and south-facing slope aspects; numbers = years since burned.*

CHAPTER II

POSTFIRE CHARACTERISTICS OF SCRUB HABITAT IN THE EASTERN DECIDUOUS FOREST-SOUTHERN PLAINS ECOTONE

ABSTRACT--The eastern deciduous forest-southern plains ecotone contains fruticose (multi-stemmed) oak (*Quercus* spp.) scrublands that are maintained in part by fire. This scrub habitat has been reduced by land-use practices and alteration of fire regimes, yet little is known of its fire ecology. We described structure and composition of scrub habitat in unburned and different-age burn treatments on the Wichita Mountains Wildlife Refuge, Oklahoma, and determined short-term postfire habitat dynamics. We found high frequencies (31-62%) of eastern redcedar (*Juniperus virginiana*) and low frequencies (30-55%) of oak (*Q.marilandica* and *Q. stellata*) in the unburned treatment, compared to 2-12% cedar and 57-89% oak in burned treatments. A relatively long (≥17 years) fire return interval was sufficient for controlling eastern redcedar in burned treatments. The lack of differences between older and newer burns suggested that habitat underwent rapid short-term (≤3 growing seasons) postfire regeneration followed by long-term stabilization. These dynamics were due to the initial prolific sprouting response of blackjack oak (*Q. marilandica*) to fire, and subsequent longer-term limitation of habitat development by arid climate and edaphic and topographic factors that probably produced low levels of soil moisture and nutrients.

INTRODUCTION--The transition between eastern deciduous forest and the southern plains is a mosaic of grasslands, forests, and savannas (Dyksterhuis, 1948; Rice and Penfound, 1959). This southwestern extreme of the deciduous forest also contains scattered scrublands in which oaks (*Quercus* spp.) have fruticose (multi-stemmed) rather than arborescent habit. Like other components in the ecotone, the scrub habitat is probably a product of fire and climatic, edaphic, and topographic factors (Buechner, 1944; Hale, 1955; Graber, 1957; Rice and Penfound, 1959; Buck, 1964; Penfound, 1968; Johnson and Risser, 1973). Although some aspects of the forest-grassland ecotone have been studied (Anderson, 1983), little is known about the scrub component or its fire ecology. The need for such information became especially critical recently when the black-capped vireo (*Vireo atricapillus*), a species whose breeding range is restricted to scrub habitat in the southern plains, was listed as endangered because of habitat loss (Ratzlaff, 1987). Listing of the black-capped vireo by the United States Fish and Wildlife Service in 1987 stimulated additional research into this ecotone.

Black-capped vireo nesting habitat contains patches of scrub oaks that have foliage within 3 m of the ground (Graber, 1957; U.S. Fish and Wildlife Service, 1991). Black-capped vireos previously nested from Kansas through Oklahoma and Texas to Coahuila, Mexico, but now are extirpated or declining in more than 50% of this area (Graber, 1957; U.S. Fish and Wildlife Service, 1991). This suggests that the distribution of scrub habitat in this region has been significantly reduced. Habitat has been destroyed directly by land-clearing and livestock browsing and indirectly by alteration of fire regimes (Graber, 1957; U.S. Fish and Wildlife Service, 1991). In addition, in the absence of disturbance gradual changes in structure or composition may alter habitat. However, fire appears to regenerate scrub habitat by decreasing the size and density of woody plants, stimulating reproductive sprouting in oaks, and reducing invading species such as eastern redcedar (*Juniperus virginiana*) (Graber, 1957; Penfound, 1968; Hayden and Tazik, 1991). Black-capped vireos nest most often on areas that have recently burned (Graber, 1957; Benson and Benson, 1990; Grzybowski 1993; J. Cornelius, personal communication).

Graber (1957), Marshall et al. (1985), and Grzybowski et al. (1994) have described aspects of scrub habitat in Oklahoma and Texas. These studies assume that fire maintains this habitat type, but there has been almost no documentation of fire effects. The only fire study was by Penfound (1968), who described tree responses to a wildfire in the Wichita Mountains, Oklahoma. Both scrub habitat and fire are still common in the Wichita Mountains. Our objectives were to describe structure and composition of scrub habitat on the Wichita Mountains Wildlife Refuge and determine short-term postfire habitat dynamics. To accomplish this, we tested the following null hypotheses: (1) there are no differences in the characteristics of burned and unburned treatments and (2) there are no differences in the characteristics of different-age burn treatments.

METHODS--Study sites were in the 24,000-ha Wichita Mountains Wildlife Refuge, which contains Precambrian granite and gabbro remains of an ancient mountain range. Soils on the sites are Stony Rock Land, Rock Land, and Granite Outcrop (U.S.D.A. Soil Conservation Service, 1967). Elevations range from 347 to 756 m. Average annual precipitation is 808 mm, but it varies greatly, and drought periods are common (Wichita Mountains Wildlife Refuge weather records). The Refuge encompasses a patchy mosaic of mixed-grass prairies, scrub oaks (*Q. marilandica* and *Q. stellata*), hardwood forests, rocky escarpments, and talus slopes (Buck, 1964; Crockett, 1964). Fires may occur in any season; prescribed fires are conducted by Refuge personnel, and wildfires are ignited by visitors and activity on the adjacent Fort Sill military reservation.

We installed 106 permanent line transects (Cain and Castro, 1959) at randomly selected points in an unburned treatment and treatments burned 1, 2, 5, and 15 years prior to the study. The 5-year-old burn treatment was divided into north and south-facing slope aspects. The 15-year-old burn treatment was divided because part of it (site B) contained scrub oak savannah, habitat notably different than the rest of the treatment (site A) (Table 1). Transects were X-shaped and consisted of a central steel post stamped at the top with an identification number, with steel stakes at each of the 4 endpoints. Transect axes were oriented along the cardinal directions, and each transect had a total length of 100 m. We sampled 100 consecutive points at 1-m intervals on the line, proceeding from south to north for points 1 to 50 and from east to west for points 51 to 100.

Transects were sampled for two consecutive years. From mid-June to early August, 1993 and late June to late July, 1994, we collected data at each point by recording contacts with an extendable 7.5-m graduated range pole placed vertically on the ground, as modified from Mills et al. (1991). Ground cover was recorded at the pole base; one canopy cover contact was recorded in each dm-interval within 2 m of the ground, as well as in each subsequent 0.5-m height interval. Cover above 7.5 m was recorded as present if an imaginary extension of the pole contacted vegetation. When more than one contact occurred in an interval, the highest contact was recorded. Abiotic components were categorized as bare ground, gravel, rock, dead wood, and grass, sedge, forb, or tree litter (Rotenberry and Wiens, 1980). Vascular plants were recorded by scientific name (U.S.D.A. Soil Conservation Service, 1982); herbaceous plants that were unidentifiable because of phenological stage were recorded as unidentifiable grass, sedge, or forb. Microphytes were categorized as lichen, moss (Bryophyta and Lycophyta:*Selaginella* sp.), and cyanobacteria.

We calculated mean values for structural characteristics of the woody species canopy in each treatment. These include absolute frequency of canopy cover (expressed as percent), height, and vertical density (Rotenberry and Wiens, 1980) of contacts in the first 2 m above the ground. We also calculated several heterogeneity indices. A vertical heterogeneity index, indicating canopy height variation, was calculated as the mean of transect standard deviations of canopy height. Two variations of a horizontal heterogeneity index, indicating patchiness or variation in the horizontal distribution of cover types, were modified from Grzybowski et al. (1994). These were calculated using the highest interval contact at each point along the line. The first was a count of the number of horizontal changes among 3 cover type categories (woody species, herbaceous species, and other, which includes abiotic components and microphytes). The second was a count of the number of horizontal changes between 2 cover type categories (woody species and the other 2 cover types combined). In addition, we calculated the relative frequency (expressed as percent) of 3 principal woody species at canopy height intervals up to 3 m.

For each structural variable, differences among treatment means were determined by 1-way analysis of variance (ANOVA), using a significance level of $\alpha = 0.05$ (Freund and Wilson, 1993). For variables having unequal treatment variances, as determined by Levene's test (Snedecor and Cochran, 1980), ANOVA was performed on rank-transformed data (Conover and Iman, 1981). Post hoc comparisons were made by Tukey's procedure (Freund and Wilson, 1993), using Kramer's approximation to account for the unequal sample sizes (SAS Institute, Inc., 1985). We used SAS computer software to perform the analyses (SAS Institute, Inc., 1985).

RESULTS-- Because of the variation within treatments, structural characteristics of the unburned treatment were not significantly different than burned treatments (Table 2). However, the unburned treatment had the greatest woody canopy height (3.3-3.4 m), fewest cover type changes between 2 categories (9-10/transect), and low frequency of woody canopy cover (28-30%), which indicated taller trees and fewer woody patches than burned treatments.

The most notable difference between unburned and burned treatments was in woody plant species composition (Fig. 1). In all treatments, grass species were dominant within the first 0.5-m of the ground. Above that, the burned treatments had high frequencies (57-89%) of blackjack oak (*Q. marilandica*) and post oak (*Q. stellata*) combined and low frequencies (2-12%) of eastern redcedar. Conversely, the unburned treatment had low frequencies (30-55%) of oak and high frequencies (31-62%) of eastern redcedar.

Significant differences occurred among burned treatments for 3 of the 6 structural variables (Table 2), but there were no trends associated with years since burned. The most notable differences were in the 2-year-old burn and site B on the 15-year-old burn. In both 1993 and 1994, the 2-year-old burn had significantly lower woody canopy height, lower standard deviation of height, and more cover type changes among 3 categories than the north-facing 5-year-old burn. In 1994, it also had significantly more cover type changes than the 1-year-old and south-facing 5-year-old burn treatments.

In both 1993 and 1994, site B (scrub oak savannah) on the 15-year-old burn had significantly lower woody canopy height than north and south-facing aspects of the 5-year-old burn, and significantly more cover type changes among 3 categories than the north-facing 5-year-old burn. In 1993, it also had significantly lower woody canopy height and more cover type changes among 3 categories than the 1-year-old burn, and lower standard deviation of height than the north-facing 5-year-old burn. Site B on the 15-year-old burn had the lowest frequency of woody canopy cover (22%) and the lowest vertical density in the 1-2 m height interval (27-30 contacts/transect) of all treatments. These data indicated that this treatment, and to a lesser degree the 2-year-old burn, contained open habitat with widely spaced, short woody patches.

DISCUSSION--Eastern redcedar is an aggressive invader in the absence of fire (Arend, 1950; Blewett, 1986; Stritzke and Bidwell, 1990). Compared to burned treatments, we found high frequencies of eastern redcedar and low frequencies of oak in the unburned treatment. Although there are no prior data describing the unburned site, it is probable that lack of fire also influenced habitat structure. Woody species canopy height was taller than burned treatments, and without fire stimulus there probably was no reproductive sprouting from oak stems and rhizomes. Fire appeared to play a vital role in maintaining scrub structure and species composition in the Wichita Mountains. Interestingly, the 15-year-old burn treatment, which was in its 17th growing season at the end of the study, had low frequencies of eastern redcedar, which indicated a relatively long fire return interval was sufficient for controlling this species in burned treatments.

It was not possible to detect habitat changes in individual treatments over the short 2-year study period, especially with the between-year weather variations occurring at the time. Extremely hot, dry weather occurred in May and June at the peak of the 1994 growing season (Wichita Mountains Wildlife Refuge weather records). This may have affected density and other variables in 1994 as grasses decreased and some trees shed foliage and entered dormancy in July. In addition, treatments on different sites and experiencing different fires may not be comparable.

Nonetheless, the lack of differences between older and newer burns suggested that habitat underwent rapid short-term postfire regeneration followed by long-term stabilization. The treatment that burned 1 year prior to the start of the study was in its 3rd postfire growing season at the end of the study. At that time it was not structurally different than the 5-year-old burn treatments or site A on the 15-year-old burn. The 2-year-old burn in its 3rd postfire growing season was not structurally different from site B on the 15-year-old burn directly adjacent to it. Although the north-facing 5-year-old burn had taller woody canopy and larger woody patches in 1994 than site A on the 15-year-old burn, the 5-year-old burn treatments were structurally similar to site A on the 15-year-old burn. In addition, black-capped vireos can be used as indicators of scrub habitat, and vireo territories damaged in the 5-year-old burn were re-occupied early in its second postfire growing season (Grzybowski, 1992). Lastly, after at least a decade of use, black-capped vireos still nested on site A on the 15-year-old burn through its 17th postfire growing season, which suggested that this area was relatively stable.

The rapid postfire regeneration of habitat within 3 growing seasons was primarily due to the initial prolific sprouting response of blackjack oak to fire. Subsequent longer-term limitation of woody vegetation was probably caused by environmental conditions. The shallow, rocky soil and steep topography of the Wichita Mountains, unique to the region, combined with arid climate to slow change in scrub habitat on slopes. Site B (scrub oak savannah) on the 15-year-old burn was an extreme example of this. It had not developed the habitat found in the rest of the burn but instead was similar to the adjacent 2-year-old burn. The area containing site B on the 15-year-old burn and the 2-year-old burn may have had low levels of soil moisture and nutrients because of shallow soil, which severely restricted woody vegetation growth and prevented development of dense scrub habitat. Reich and Hinckley (1980) suggested that a pygmy oak ecosystem in Missouri was produced by low soil nutrients and secondarily by low soil moisture.

Long-term monitoring will be necessary to determine the lifespan of scrub habitat in the Wichita Mountains. In addition, fires in our study treatments occurred in the dormant season; growing season fires affect habitat differently (Adams et al., 1982; Huffman and Blanchard, 1991). A growing season fire occurring during drought, when photosynthesis is reduced, could have especially severe effects (Johnson and Risser, 1973). During extended drought in the 1950's, Graber (1957) found that tree death resulted in loss of scrub habitat in Kansas, Oklahoma, and Texas. The stress of producing foliage in response to growing season fire during drought might kill trees by depleting carbohydrate reserves (Aldous, 1934; Garrison, 1972) and cause habitat to shift from scrub woodland to savannah or grassland.

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TABLE 1--Summary of treatments studied during 1993 and 1994 on the Wichita Mountains Wildlife Refuge, Oklahoma.

Code	No. Transects	Description
1	20	Burned February, 1992, one year prior to study; 1,027 ha prescribed fire.
2	13	Burned March, 1991, two years prior to study; 851 ha wildfire.
5N	20	North-facing aspect. Burned December, 1988, five years prior to study; 1,581 ha wildfire.
5S	20	South-facing aspect. Burned December, 1988, five years prior to study; 1,581 ha wildfire.
15A	20	Site A. Burned March, 1978, fifteen years prior to study; >6,070 ha wildfire.
15B	7	Site B (scrub oak savannah). Burned March, 1978, fifteen years prior to study; >6,070 ha wildfire.
NOBURN	6	Unburned. Possibly burned March, 1959; 111 ha wildfire. Ten transects were sampled both study years; 4 were deleted from analyses after burning in April, 1994; 931 ha wildfire.

TABLE 2--Mean values for structural characteristics of woody species canopy in scrub habitat on the Wichita Mountains Wildlife Refuge, Oklahoma.[1]

	Woody Canopy (%)		Height (m)		Vertical Density (No. contacts) Height interval 0-1 m		Vertical Density (No. contacts) Height interval 1-2 m		Std. Deviation of Height (m)		No. Cover Type Changes Among 3 Categories		No. Cover Type Changes Between 2 Categories	
Treatment[2]	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994
1	36	36	2.4abc	2.4abc	62	62	49	47	1.1a	1.2ab	30ab	31ab	16	16
2	32	32	1.6acd	1.5ac	86	84	46	35	0.9a	1.0a	36ac	40c	18	20
5N	48	47	3.0b	2.9b	72	65	51	48	1.5b	1.5b	27b	26b	17	17
5S	43	41	2.3bc	2.3bc	91	90	57	53	1.3ab	1.4ab	31abc	32ab	17	18
15A	35	33	1.9acd	2.0ac	72	61	63	50	1.1a	1.1ab	33abc	36ac	16	17
15B	22	22	1.2d	1.3a	74	65	27	30	0.8a	1.0ab	41c	40ac	18	18
NOBURN	28	30	3.3abcd	3.4abc	67	63	47	59	1.0ab	1.1ab	31abc	34abc	9	10

[1] Means (within column) followed by different letters are significantly different ($\alpha = 0.05$).
[2] Numbers = years since burned; N and S = north and south-facing slope aspects; A = site A; B = site B (scrub oak savannah).

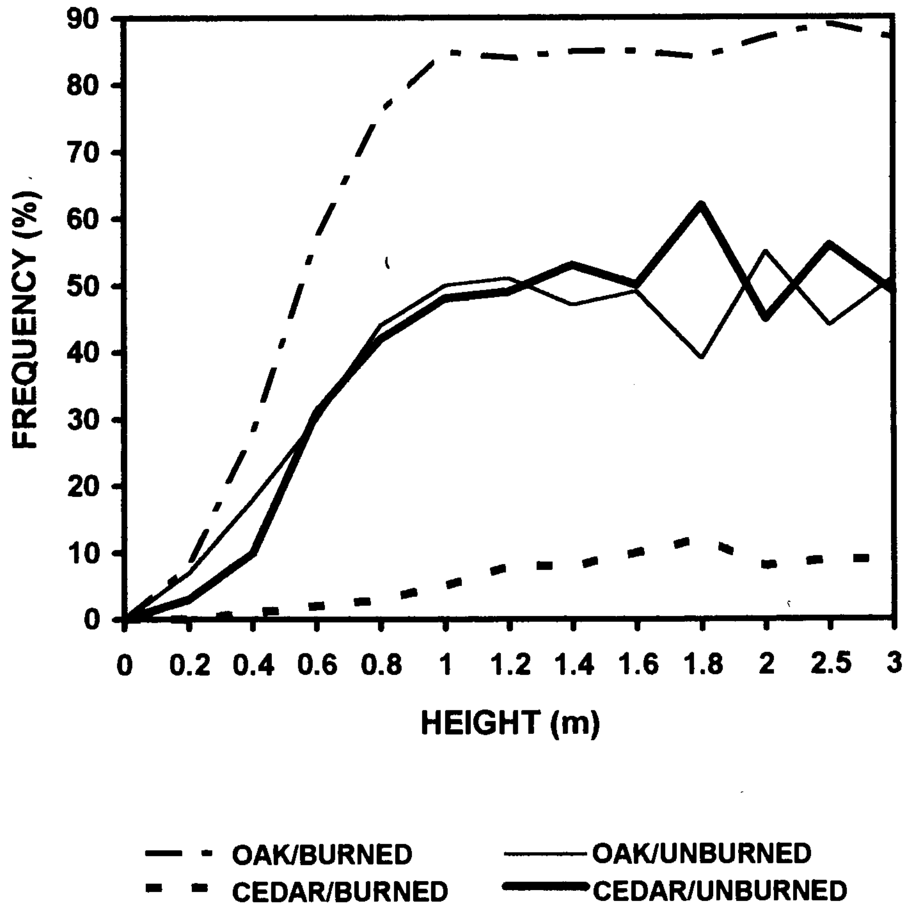

Fig.1--Relative frequency of oak (blackjack and post oak combined) and eastern redcedar canopy cover in burned and unburned treatments. Values are means of 1993 and 1994 data and burned category values are means of all burned treatments.

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