Population Status and Threat Analysis for the Black-capped Vireo

Neal Wilkins, Robert A. Powell, April A.T. Conkey
and Amy G. Snelgrove
Department of Wildlife and Fisheries Sciences
Texas A&M University

Prepared For
U.S. Fish & Wildlife Service, Region 2

May 2006

With contributions from

Linda Campbell, David Cimprich, Craig Farquhar, Pat Reardon, Don Petty, Duane Schlitter, Douglas Slack and David Wolfe

CONTENTS

LIST OF FIGURES

1.0 Introduction

The black-capped vireo (BCVI; *Vireo atricapilla*) is a migratory bird with a known breeding occurrence throughout portions of central Texas, the state of Coahuila in Mexico, and isolated areas in Oklahoma. The former breeding range includes a portion of north-central Texas, most of central Oklahoma, and south-central Kansas. The bird's wintering range is on the Pacific slope in western Mexico. On November 5, 1987, the species was listed as Endangered under the Endangered Species Act (ESA) of 1973, as amended. The primary threats supporting the decision to list the species were habitat loss from development; habitat destruction from the grazing of sheep, goats and exotic livestock; and nest parasitism by brown-headed cowbirds (*Molothrus ater*), as determined by the U.S. Fish and Wildlife Service (USFWS) (Ratzlaff 1987). A recovery plan was developed in 1991 (USFWS 1991), but critical habitat has not been designated, and a status review has not been conducted since the listing.

The USFWS initiated the review process for the BCVI in February 2005 by issuing a notice of review and request for information on the species. The purpose of the scientific review effort is to evaluate all scientific and commercial information available on the present status of the BCVI. This evaluation will provide the USFWS with the data needed for making determinations under a status review as required by Section 4(c)(2) of the ESA.

1.1 Objectives

According to Section 4(c)(2) of the ESA, the purpose of a 5-year status review is to assess the following: (a) whether the present population appears to be declining, stable or increasing since the time of listing; (b) whether the threats identified at listing are increased, unchanged, reduced or eliminated; and (c) whether there are any new threats to the species.

1.2 Approach

The overall approach taken was to accumulate, summarize and evaluate the existing information on the species. This information is in the form of peer-reviewed scientific literature, published reports, expert opinion, unpublished manuscripts, archives of published and unpublished data, and a variety of public records. No new data was collected under this effort, and new analyses were not conducted beyond the basic and summary statistics required to gain a range-wide perspective on the central issues addressed in the status review. Where existing data were not adequate for reaching reliable conclusions, that fact is stated and the apparent gaps in information noted.

This report does not make any recommendations concerning the listing status of the species or changes in the listing status. This remains the responsibility of the USFWS.

1.2.1 Review panel

An eight-person review panel was assembled to assist in identifying relevant information and to provide review during the information collection and evaluation process.

The project principal investigator and staff selected review panel members, who included land managers, wildlife biologists and other scientists with expertise appropriate to one or more of the issues being evaluated. Members of the review panel are:

Linda Campbell
Texas Parks and Wildlife Department
4200 Smith School Road
Austin, Texas 78744
Linda.campbell@tpwd.state.tx.us

David Cimprich
The Nature Conservancy
P.O. Box 5190
Fort Hood, TX 76544-0190
dcimprich@tnc.org

Craig Farquhar
Texas Parks and Wildlife Department
3000 S. Interstate 35, Suite 100
Austin, Texas 78704
craig.farquhar@tpwd.state.tx.us

Doug Slack
Texas A&M University
2258 TAMU
College Station, Texas 77843-2258
d-slack@tamu.edu

Don Petty
Texas Farm Bureau
P.O. Box 2689
Waco, Texas 76702-2689
dpetty@txfb.org

Pat Reardon
Texas Wildlife Association
P.O. Box 1661
Mason, Texas 76856
Patrick.reardon5@verizon-net

Duane Schlitter
Texas Parks and Wildlife Department
3000 S. Interstate 35, Suite 100
Austin, Texas 78704
duane.schlitter@tpwd.state.tx.us

David Wolfe
Environmental Defense
44 East Avenue
Austin, Texas 78701
dwolfe@environmentaldefense.org

The review panel helped the project staff identify data sources and interpret the data collected. Panel members developed preliminary assessments in their areas of expertise and provided critical review of the evaluations and conclusions contained in the final report. The panel met several times in the course of this project.

1.2.2 External peer review

The status review process calls for the principal investigator and project staff to seek external peer review of the draft final report. Thus, we sought 3 external reviews of our January 2006 “Final Draft” from wildlife researchers recommended by the review panel and other wildlife professionals. The detailed reviews were provided to USFWS personnel, and this document includes revisions as suggested by those reviews. We acknowledge the contributions from the critical external reviews provided by J.D. Brawn, D. Buehler, and M. Morrison.

2.0 Life History

2.1 Introduction

This section describes the basic life history of the BCVI, including a summary of the species' documented geographic range, food habits, reproduction and mortality.

2.2 Species Description

The BCVI is a small, migratory songbird 10 to 12 cm long (Graber 1957, Grzybowski 1995, Howell and Webb 1995). It is unique among vireos in being sexually dichromatic (Graber 1957) and in showing delayed plumage maturation in first-year males (Rohwer et al. 1980). Mature males are mostly olive green above and white below with faint greenish-yellow flanks (Oberholser 1974, Campbell 1995). The crown and upper half of the head are black, and the partial white eye-ring connects with white lores to form "spectacles." The bill is black, and the iris is red in mature males and brownish red or amber in females and immatures (Graber 1957, Howell and Webb 1995, Pyle 1997). The plumage of females is duller overall than that of males. The heads of females are dark slate gray (USFWS 1991, Campbell 1995, Grzybowski 1995).

2.3 Geographic Range

In 1986, the known breeding range of the BCVI included portions of Kansas, Oklahoma, Texas and central Coahuila, Mexico (Shull 1986). Today, the breeding range no longer appears to extend farther north than central Oklahoma, but it apparently extends farther south than was previously known (Farquhar and Gonzalez 2005). The information collected for this status assessment indicates that the boundaries of the breeding range should be modified to exclude Kansas and extend southward through the Mexican state of Nuevo Leon and into the southwestern part of Tamaulipas (Fig. 2.1). The information collected for the succeeding sections supports this description of the bird's present range. Since its listing, the species has not been known to occur outside this range. The wintering range for the species is not as well documented but appears to be along the Pacific coast of Mexico from approximately 27 degrees to 16 degrees latitude (Fig. 2.1). For reference throughout this document, the species' U.S. breeding range is divided into geographic units (Fig. 2.1) as suggested by the Population and Habitat Viability Assessment Report (USFWS 1996).

Breeding range in Mexico. Prior to recent observations by Farquhar and Gonzalez (2005), the species was not confirmed to breed farther south than central Coahuila, Mexico (Renardo 1886, Moore 1938, Miller 1955, Van Hoose 1955, Graber 1961, Wauer and Ligon 1977, Benson and Benson 1990). Until recently, the only evidence of breeding south of Coahuila was unconfirmed single records from Tamaulipas (Phillips 1911), San Luis Potosi (Davis *in* Graber 1961), and Nuevo Leon (compiled in Marshall et al. 1984 and Marshall et al. 1985). Recent records document breeding in Nuevo Leon (in Bustamante; the first confirmed Mexican breeding records outside of Coahuila; Farquhar et al. 2003) and in southwestern Tamaulipas (Palmillas; Farquhar and Gonzalez 2005). The Tamaulipas records (approximately 20 adult BCVIs and four fledglings) are at least 700 km south of the previous confirmed southern records from Coahuila; they are also the first documented breeding records for this species south of the Tropic of Cancer (Farquhar and Gonzalez 2005). Based on these recent reports from northeastern Mexico, the known breeding range in Mexico has been extended southward, producing a distribution map as seen in Figure 2.1.

Winter range in Mexico. The known non-breeding, winter range consists of an elongated and patchily distributed area along the Pacific slopes of the Sierra Madre Occidental Mountains in Mexico, extending from southern Sonora to Oaxaca (Fig. 2.1). Marshall et al. (1985) discussed the winter range as including the Mexican states of Sonora, Durango, Sinaloa, Nayarit, Jalisco, Michoacan, Guerrero, Oaxaca and possibly Hidalgo. There is no evidence that BCVIs winter in Hidalgo (the lone record was of a probable migrant in mid-October [Marshall et al. 1985]), but the other states listed by Marshall et al. (1985) are accurate, although one additional state where the species is known to winter (Colima) is absent from their list. Mexican states where occurrence has been documented include Sonora (Russell and Morrison 1996), Sinaloa (Graber 1957, Graber 1961, Marshall et al. 1985, Howell 1999, Powell unpublished data), Durango (Graber 1957, Howell and Webb 1995, Powell unpublished data), Nayarit (Marshall et al. 1985, Howell 1999, Powell unpublished data), Jalisco (Hutto 1989, Hutto 1994, Howell 1999, Powell unpublished data), Colima (Howell 1999, Powell unpublished data), Michoacan (Howell and Webb 1995), Mexico (probable migrant; Escalona et al. 1995), Guerrero (Howell and Webb 1995), and Oaxaca (Binford 1989, Howell 1999). With the exception of Sonora, there are also specimen records from the same states (Appendix B).

Most of the non-breeding records are concentrated in Sinaloa and Nayarit, and this area has been described as the center of the wintering grounds (Graber 1961). However, Sinaloa and Nayarit were the states most heavily surveyed during previous studies, so the large proportion of records in those states may largely be an artifact of sampling effort. Recent research indicates that the birds may be most heavily distributed in the states of Sinaloa, Nayarit, Jalisco and Colima (Powell, unpublished data).

Migration. Moore (1938) commented that the scarcity of BCVI records from Sonora suggests that the birds cross the tableland through Chihuahua and descend through the canyons of southwestern Chihuahua and Durango to the coast. Graber (1961) agreed with the idea of such a route, but she visited the area and found no evidence of habitat suitable for BCVIs. Marshall et al. (1985) and Farquhar and Gonzalez (2005) also doubted the likelihood of this migratory route because of the mountainous terrain and the xeric conditions along the way, even though it would be the shortest linear route between the wintering areas and the breeding grounds in Texas and Oklahoma. Graber (1961) stated that scattered records of BCVIs from high elevations (one as high as 9,500 feet) suggest the possibility of a migratory route over the mountains. Moore's (1938) proposal was made prior to the discovery of the currently known breeding range in Mexico. Farquhar and Gonzalez (2005) thus suggested that these southern populations might instead migrate across the shrubby, submontane vegetation associated with the Volcanic Belt Pine-Oak Forests. Similarly, Marshall et al. (1985) examined specimen and site records of BCVIs during migration and proposed the possibility of a route around the edge of the plateau to the south, along or parallel to the Sierra Madre Oriental.

Black-capped vireos begin to depart from the breeding grounds in late August and September, with the young birds leaving first, followed by the adult females and then the adult males (Graber 1961, Marshall et al. 1985). In the spring, they arrive on the breeding grounds about a week after the average date of the last frost (Graber 1961), which is usually from mid-March to mid-April in Texas and approximately 10 days later in Oklahoma (Campbell 1995, Grzybowski 1995). Males typically arrive about a week or two before females and first-year males to select their territories (Graber 1961, Campbell 1995).

2.4 Nesting and Reproduction

Nesting begins shortly after the females arrive on the breeding grounds (Graber 1961). Males sing to attract mates and defend territories, which usually range in size from 1 or 2 hectares (ha; mean=1.5; Graber 1961) to 10 ha (mean=3.6 ha; Tazik 1991b). Pairs form after a brief courtship (less than 1 to 2 days). Pairs remain socially monogamous throughout the breeding season and select nesting sites together (Grzybowski 1995). It takes 2 to 9 days for females to construct the cup-shaped nests, which are suspended in the forks of shrubs in dense underbrush from 0.2 to 3.0 m (usually 0.5 to 2.0 m) above the ground (Campbell 1995, Grzybowski 1995).

The first egg is usually laid 2 days after nest completion; additional eggs are laid on each subsequent day (Graber 1961). The first nesting attempt usually results in three to four eggs, while later clutches may only contain two to three eggs (Campbell 1995). Incubation takes 14 to 17 days, with both males and females sharing incubation duties. Likewise, both males and females share the responsibility of feeding the chicks, which leave the nest 10 to 12 days after hatching (Campbell 1995).

2.5 Food Habits and Foraging Behavior

Black-capped vireos are opportunistic gleaners of insects (Graber 1961). Their spring and summer diet consists primarily of insect larvae, most of which are of the Order Lepidoptera (Graber 1961, Grzybowski 1995). The stomach contents from eight BCVIs captured range-wide during the breeding season was composed of Arthropods (94.1 percent) and seeds (5.9 percent), with major items including Lepidoptera adults (16.2 percent) and larvae (13.2 percent), Coleoptera (30.9 percent), Homoptera (10.3 percent), and Arachnida (Araneida; 7.4 percent) (Graber 1957, Graber 1961). The fall and winter diet appears to include a wider array of insect matter, as well as vegetable matter (primarily seeds, but possibly fruits as well), although data on this subject are very limited (Graber 1961, Powell personal observation). These data only supply limited information upon which to base management.

Both males and females forage in woody vegetation at all levels and seem to prefer deciduous substrates (especially oaks) (Graber 1961, Grzybowski 1995). Foraging behavior typically involves gleaning from leaves, twigs and small branches, as well as from the trunks of trees (Grzybowski 1995).

2.6 Survivorship and Mortality

Estimates of adult annual survivorship are variable. Based on returns of color-banded birds at Kerr Wildlife Management Area (WMA), the annual survivorship of adult males was estimated at 0.55 to 0.75 (Grzybowski 1991). Estimates of annual survival rates for adult males at Dolan Falls, Texas was 0.68 (*n*=63), while at Fort Hood, Texas it was 0.40 (*n*=884) (Alldredge et al. 2003). Population viability analysis (PVA) models for BCVI have used values of 0.57 for annual adult survival rates (USFWS 1996, Parysow and Tazik 2002). The above estimates were all derived from return rates, and should therefore be considered as minimum survival rates.

Adult female survivorship is not well established, but is likely to be lower than for males because sex ratios favor males (Grzybowski 1995). Juvenile survivorship also is not well established, but it has been suggested by Grzybowski (1995) to be higher than that of adults. Data of Alldredge et al. (2003) suggested hatch year survival rates of 0.17 (*n*=822), but these results may be due to high natal dispersal, which is a common trait of neotropical migrants (Villard et al 1995). Population viability analysis (PVA) models for BCVI have used values of 0.43 for juvenile survival rates (USFWS 1996, Parysow and Tazik 2002).

The most common predators of free-flying BCVIs are snakes and accipiters, although little data are available on this (Grzybowski 1995). Nests are frequently lost to predation. Of 225 eggs at sites in Oklahoma and Texas, 31 (14 percent) were believed lost to predators (Graber 1961). At Fort Hood (where cowbird removal has reduced nest loss from parasitism and subsequent nest abandonment), depredation was the largest cause of nest failure in 2004, accounting for 82 percent of the unsuccessful nests (*n*=166) and 44 percent of all nests (*n*=312) (Cimprich 2004). At Fort Hood, a nest-monitoring study from 1998 to 2001 found that snakes and fire ants (*Solenopsis* spp.) were the leading predators, accounting for 38 percent and 31 percent, respectively, of all depredated nests (*n*=48) (Stake and Cimprich 2003). Other nest predators in the Fort Hood study included avian predators (19 percent of depredated nests) and mammalian predators (11 percent). The fact that predation appears to increase when cowbird parasitism is decreased through removal programs suggests that predation by snakes and fire ants might limit some populations when cowbird parasitism is reduced.

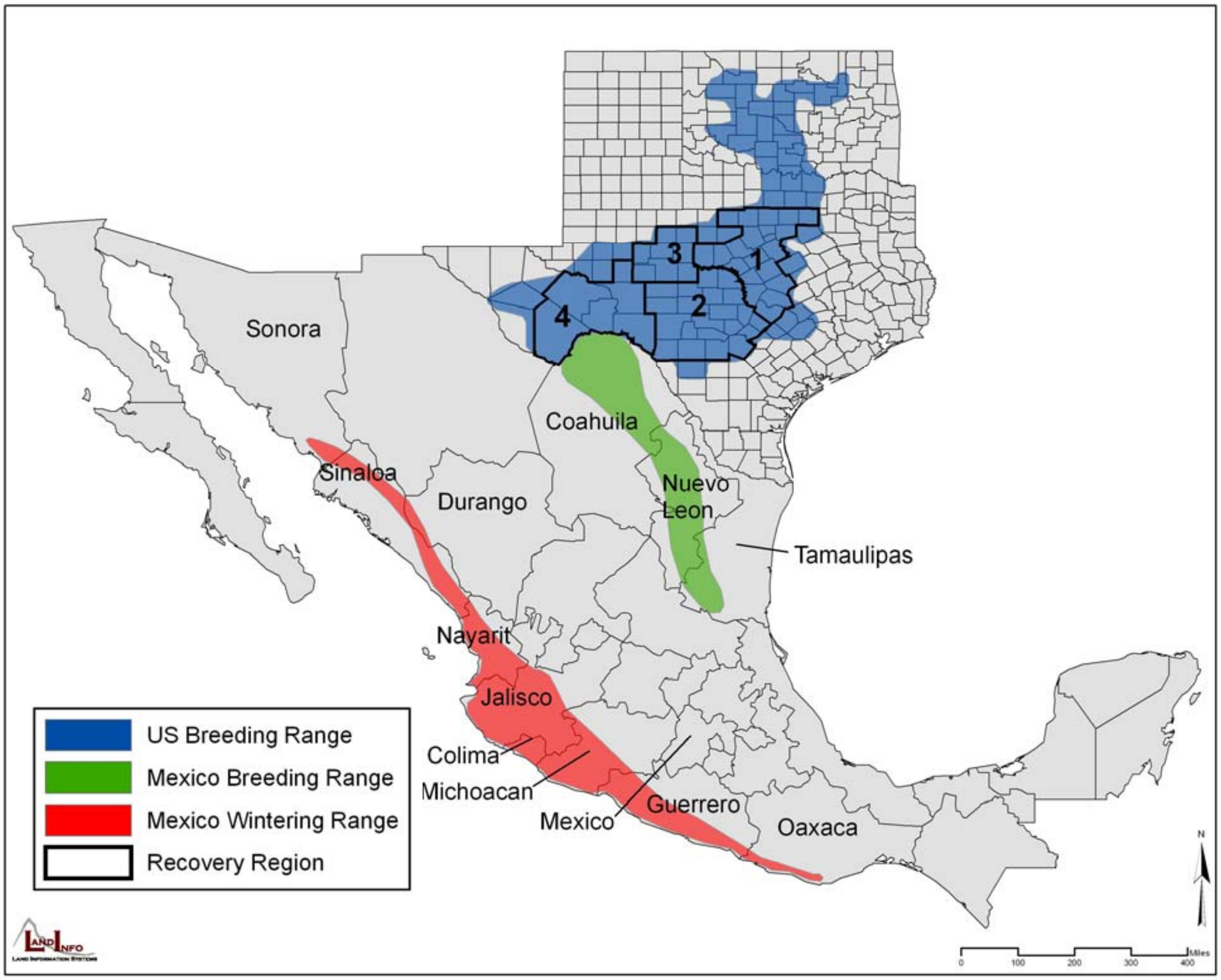

Figure 2.1. Currently known breeding and wintering ranges for the Black-capped Vireo with recovery regions as suggested for revision by the Population and Habitat Viability Assessment Report (USFWS 1996. Ranges are generalized for all known locations since time of listing. Note:Black outlines in the Texas range of the species represent recovery units.

3.0 Habitat Characteristics and Availability

3.1 Introduction

Here we discuss general characteristics of BCVI breeding habitat, nest site characteristics, and factors influencing habitat suitability, including the role of fire. Following that are estimates of the amount of habitat suitable for BCVIs in Texas and Oklahoma and an overview of known BCVI habitat relationships on the breeding and wintering grounds in Mexico. More information is available for breeding habitat than for wintering grounds.

3.2 Limitations

Ideally, the characteristics and availability of habitat should be considered at various scales: the landscape scale, the patch scale, and nest location. The monitoring of habitat suitability at the landscape scale would provide a metric for assessing the conservation status of the species. However, landscape-scale habitat information for the BCVI is severely limited. While there is much more information at the habitat patch and nest location scales, its utility is largely confined to site-specific management implications and it is not of much direct use in determining the status of the species.

3.3 Habitat Availability at the Landscape Scale

Estimates of the amount of suitable habitat at the landscape scale are generally made with the aid of remotely sensed data and data-intensive GIS analyses. The use of habitat modeling and remote sensing to construct habitat suitability models for BCVIs has been limited, partly because of the difficulty of distinguishing canopy-to-ground foliage cover, which is necessary for identifying suitable BCVI habitats. Recently, Light Detection and Ranging (LIDAR) was used at Fort Hood to describe and locate potential BCVI habitat patches based on size, shape, and vertical structure of vegetation (Leyva et al. 2004). Overall accuracy in classifying habitat (BCVI habitat and non-habitat) in this preliminary study was only 69% (*n*=199). Although this methodology is limited by expense and availability, its use may hold promise for future work.

The only wide-ranging estimates of suitable habitat for the species come from a recent USFWS Biological Opinion for brush management in Texas (USFWS 2004). The assessment relied heavily upon roadside surveys of 53 Texas counties conducted between July 1996 and August 1998 (Maresh et al. 1999, Maresh and Rowell 2000). In each of these counties (except Dallas County), two 30-mile transects were surveyed for BCVIs and estimates were made of the potential suitable habitat within the survey transects. The estimates of potential habitat within each county were then made by extrapolating the estimates from roadside surveys with USGS topographic maps. One exception to this was Dallas County, where no roadside surveys were conducted. Instead, potential habitat was estimated from an assessment of USGS topographic maps of areas of known occurrence within the county.

In three other counties (Montague, Brewster and Pecos), no roadside habitat segments were recorded (Maresh et al. 1999, Maresh and Rowell 2000), so potential habitat was estimated from an examination of topographic maps and recent site visits (USFWS 2004). Table 3.1 presents the habitat assessments from the Biological Opinion. It includes the estimated approximate land area by county, the area of potential BCVI habitat, and the proportion of total county land area potentially suitable as BCVI habitat.

According to the habitat assessments in Table 3.1, the 53 counties inventoried contained approximately 1.45 million acres of potential suitable habitat. This is equivalent to about 3.3 percent of the total land area considered. The estimate was made to support decisions concerning the amounts of potential habitat that might be manipulated by brush management during any one year. It should be mentioned, however, that the survey routes were chosen specifically to include areas of known or "most likely" BCVI occurrence, thus limiting the statistical rigor of the study and likely resulting in overestimates of occupied and potential suitable habitat. Furthermore, this estimate did not consider variation in habitat quality or variance for determining statistical confidence, and it is not comparable to any data collected in previous years. So, while this assessment was informative for the process of forming a Biological Opinion for the incidental take associated with brush management, it is of limited value in comparing current available habitat against the available habitat at time of listing. Also, due to the lack of statistical reliability or the random selection of survey routes, these estimates should be interpreted with caution. We found no estimates of suitable BCVI habitat available for either Oklahoma or Mexico.

3.4 Patch-level Habitat Characteristics

U.S. range. In Texas and Oklahoma, suitable BCVI habitat is characterized by a patchy distribution of low, scrubby growth made up mostly of deciduous woody shrubs and trees of irregular height (Graber 1961). When compared with adjacent habitats, the habitats in BCVI territories have a higher density of deciduous vegetation less than 2 m in height (Grzybowski et al. 1994). In an analysis of habitat across Texas and Oklahoma, deciduous cover around BCVI nests was typically 30 to 45 percent across the range, while total woody cover, including junipers (*Juniperus* spp.), was 36 to 55 percent (Grzybowski et al. 1994). Similarly, Juarez (2004) found average canopy cover by low-growing (less than 1.5 m) shrubs to be 18.6 percent at sites occupied by singing males during the breeding season, which was more than double that found at sites not known to be occupied.

Where there is low-growing, deciduous cover, BCVIs are more likely to occupy areas with sparser cover by *Juniperus* spp. (Grzybowski et al. 1994, Juarez 2004). For example, on private ranch land in Coryell and Hamilton Counties in Texas, singing males were found on sites with an average juniper cover of 9.3 percent (Ashe juniper, *J. ashei*, less than 3 m high), while sites with a cover of 19.6 percent were unoccupied (Juarez 2004).

Recent work conducted at Fort Hood suggests that habitat may be a limiting factor for BCVI (Noa 2005). Black-capped vireo abundance and age structure were compared between two habitat types – shrubland habitat and scattered patches of shrubby vegetation centered on one or several large trees (resulting from armored vehicle traffic) – over a 2-year period. The latter had a lower abundance of BCVI (*n*=63) than did the shrubland habitats (*n*=115) and a higher percentage of second-year males (49% vs.32%), suggesting that young BCVI may use lower quality habitats (Noa 2005). Grzybowski et al. (1994) also reported areas of suboptimal BCVI habitat that had higher ratios of second-year males, and it is not uncommon that younger males should occupy habitat that is less than optimal (Holmes et al. 1996, Petit and Petit 1996).

Mexican breeding range. Black-capped vireo populations in northern and central Coahuila have been described as occupying a mountainous zone with scrub-oak vegetation (Marshall et al. 1985, Grzybowski 1995). Throughout this region, distinct altitude-dependant vegetation belts occur (Miller 1955, Benson and Benson 1990). Graber (1961) described the habitat in Sierra Padilla, Coahuila as similar to that described by Lesueur (1945) for isolated limestone Sierras in northeastern Chihuahua, where the lowest oaks (*Quercus* spp.) appear at

5,600 feet, junipers grow at 5,800 feet, and a few large pines (*Pinus* spp.) grow on isolated ridges. As Graber (1961) described it, the habitat in Sierra Madera in Coahuila was similar to that at Sierra Padilla, where pine forests were more prevalent at the higher slopes, and similar to the descriptions of the Sierra del Carmen by Miller (1955). Black-capped vireos were found only on dry, limestone hillsides with thick mats of vegetation 3 to 5 feet high (Graber 1961).

Benson and Benson (1990) described the lowland habitat as desert shrub that extended from the base of the mountains up to the mouths of the canyons. Above this was habitat described as "montane low forest" (Muller 1947); it was dominated by live and deciduous oaks, as well as pinyon pine (*P. monophylla*), juniper, walnut (*Juglans* spp.) and elm (*Ulmus* spp.). Benson and Benson (1990) described the elevations above 1,300 m as pine-oak woodlands (primarily Ponderosa pine; *P. ponderosa*) with dense shinnery oak (*Q. havardii*). Above 2,000 m, mesas dominated by conifers were typical. Howell and Webb (1995) described BCVI habitat in Mexico as arid to semiarid scrub, especially with oaks, and they noted that the birds commonly nest at low to mid-levels in brush or scrubby trees.

Much of the vegetation suitable for BCVIs across the Mexican breeding sites in Coahuila, Nuevo Leon and Tamaulipas states can be characterized as either Tamaulipan thornscrub or submontane pine-oak chaparral (Farquhar and Gonzalez 2005). Tamaulipan thornscrub is a xeric habitat typically found below 1,000 m where the vegetation is patchy and low-growing (less than 3 m) as in large areas of west Texas (e.g., Big Bend National Park, Dolan Falls Ranch, and Devils River State Natural Area). The persistence of BCVI habitat in Tamaulipan thornscrub habitat type does not appear to depend on fire. The submontane pine-oak chaparral is found at higher elevations and is generally associated with foothills and the lower slopes of the Sierra Madre Oriental (western slopes in Nuevo Leon and Tamaulipas). The persistence of BCVI habitat in the submontane pine-oak chaparral, (consisting of low-growing oaks, sumacs [*Rhus* spp.], junipers and sotol [*Dasylirion wheeleri*]) may depend on fire to retard secondary succession (Farquhar and Gonzalez 2005). Interspersed among these areas are rocky slopes with shallow soils that are unable to support trees with deep roots; fire is probably not necessary to maintain the habitat in these areas (Farquhar and Gonzalez 2005).

Mexican wintering range. The winter range of the BCVI has received little attention, so there is only limited information about it. Graber (1957, 1961) qualitatively assessed wintering habitat requirements, determining that they are complex and have a wider range of vegetation types than do the breeding grounds. Based on her observations (which were restricted to sites in Sinaloa and Nayarit states), birds chose both arid scrub 0.75 to 3.0 m high (southern Sinaloa) and mesic, luxurious, secondary growth with a richness of plant species (coastal Nayarit). The latter plant community was suspected to have resulted from widespread clearing; it appeared to be particularly favored by the BCVI. Howell and Webb (1995) described wintering BCVI habitat as either arid to semiarid scrub (especially where there are oaks) or humid, brushy, secondary growth and forest edge.

During research in the winters of 2002-2003 and 2003-2004, BCVIs (*n*=56) in Sinaloa, Durango, Nayarit, Jalisco and Colima states selected habitat with significantly less canopy cover, denser shrubs and steeper slopes than random habitat points (Powell, unpublished data). This research confirmed the general assessments by Graber (1961) and Howell and Webb (1995) that BCVIs selected both mesic secondary growth and xeric scrub, although they used a variety of other habitat types as well, including shade coffee plantations, thorn forest, riparian forest, pine-oak forest and deciduous forest. Black-capped vireos were also found at a variety of altitudes from sea level to 1,462 m (4,798 feet), with a mean altitude of 585 ± 101 m (*n*=56) (Powell, unpublished data).

3.5 Nest Site Characteristics

Black-capped vireos construct their nests relatively near the ground, with most nests (90 percent) occurring 0.4 to 1.25 m above ground level (Grzybowski 1986). Low-growing branches of several species of oaks (*Quercus* spp.) provide the majority of known nesting substrates (Graber 1957, Tazik et al. 1989, Grzybowski et al. 1994).

Within habitat patches, BCVI nest sites tend to be in deciduous vegetation in areas with no more than 69 percent woody cover, considerable edge density (transitions among 2 or more patch types), and heavy foliage cover below 2 m (Bailey 2005). According to Grzybowski et al. (1994), the woody canopy cover immediately adjacent to BCVI nests is 35 to 55 percent across the species range. Most of the cover is deciduous (39.7 percent deciduous cover ± 13.81) rather than juniper (7.6 percent juniper cover ± 8.66). While juniper is occasionally used for nest locations, it is generally underused relative to availability in Texas and Oklahoma (Grzybowski

1986, Tazik and Cornelius 1989, Tazik et al. 1989, Grzybowski 1995, Bailey 2005)[1]. In breeding habitats at Fort Hood, Bailey (2005) found that BCVIs were almost three times (283 percent) more likely to nest in deciduous cover than in juniper.

Blackjack oak (*Q. marilandica*), shin oak (*Q. sinuata*), Spanish oak (*Q. texana*), plateau live oak (*Q. mohriana*) and Vasey oak (*Q. pungens* var. *vaseyana*) are the most frequently used species at nesting sites in Texas and Oklahoma (Graber 1957, Tazik et al. 1989, Grzybowski et al. 1994). Other common species in Texas and Oklahoma that compose the deciduous cover in BCVI habitat include sumac, Texas persimmon (*Diospyros texana*), roughleaf dogwood (*Cornus drummondi*) and redbud (*Cercis canadensis*) (Grzybowski 1995). Other species common to BCVI habitat include Texas ash (*Fraxinus texensis*) and Mexican buckeye (*Ungnadia speciosa*). At sites in northern Mexico, common species include oaks (*Q. grisea* and *Q. laceyi* in Coahuila, *Q. invaginata* and *Q. fusiformis* in Nuevo Leon) and sumac (*R. virens*) (Farquhar and Gonzalez 2005).

3.6 Factors Influencing Habitat Suitability

Suitable habitat for the species results from the combined effects of drought, periodic fire, and grazing pressures interacting with site characteristics such as landform, topography and dominant vegetation type. Depending on climate and other physical factors, the proper conditions for breeding habitat can be relatively short-lived. Fire, and in some cases moderate browsing by wildlife and livestock, can maintain suitable successional stages for the development of breeding habitats (Ratzlaff 1987). As a consequence, BCVIs are often found in areas with recent histories of fire, and the highest BCVI concentrations typically occur in areas recovering from a hot fire (Graber 1957, Marshall et al. 1985, Grzybowski et al. 1994). Under the proper burning conditions, fires can kill or retard invading junipers and favor the regrowth of fire-adapted oak and sumac species, thus providing the areas of dense, low foliage required by BCVIs (USFWS 1991, Campbell 1995, Grzybowski 1995).

Where there is moist soil, as is commonly found in the eastern two-thirds of the breeding range, BCVI habitat changes through succession into closed-canopy hardwood forest (Grzybowski et al. 1984). Some BCVI territories, however, are located on steep slopes where the

---

[1] We found no quantified data on nest-site characteristics in Mexico.

shallow soils slow succession and the microclimate perpetuates the clumping of vegetation suitable for BCVI habitat (Graber 1961). In general, the habitats used by BCVIs in southwestern Texas and northeastern Mexico are less influenced by succession than the areas in the northern and eastern portions of the range (Farquhar and Gonzalez 2005). Some areas of Mexico (i.e., Rancho La Escondida, Coahuila) do contain deep-soiled, shrubby oak BCVI habitat adjacent to fire-dependent pine forest. Although these habitat areas are affected by the fires, it is not thought that managing them with fire would be necessary to maintain BCVI habitat (Farquhar and Gonzalez 2005).

Marshall et al. (1985) also noted that in some areas (e.g., Kerr County, Texas) browsing by white-tailed deer can actually maintain a low-growth form of preferred nest substrates such as shinnery oak. However, if white-tailed deer populations exceed an areas' "carrying capacity," the resultant overbrowsing can diminish habitat suitability for BCVIs.

Experimental work by Ward and Schlossberg (2004) at Fort Hood, produced evidence that BCVIs are attracted to specific sites by recorded vireo vocalizations. Their results suggest that BCVIs may use vocalizations as a cue in identifying areas suitable as breeding habitats, thus implying that artificial stimuli may be used as a conservation tool for the species.

Influence of fire. The absence of fire on many rangelands and woodlands has led to the degradation of much potential BCVI habitat by allowing successional advancement and the encroachment of junipers. In the Edwards Plateau of Texas, the absence of fire has contributed to the encroachment of Ashe juniper onto open woodlands (Smeins and Merrill 1988). Recurring rangeland fires were a primary influence in the development of an oak-dominated plant community throughout much of the species range; the suppression of fire contributes to juniper invasion and dominance (Gehlbach 1988). Therefore, fire is important in creating and maintaining BCVI habitat across much of the eastern and northern portions of the species' breeding range (Graber 1961, Shaw et al. 1989, Benson and Benson 1990, USFWS 1991).

Several studies have addressed the effects of fire on BCVI populations and have attempted to estimate the time interval at which BCVI occupancy or re-occupancy occurs. Black-capped vireos fully recolonized burned areas the second year post-burn at Wichita Mountains Wildlife Refuge (WR) (Grzybowski 1989, Grzybowski 1990a), and early results from Fort Hood, Texas suggested a similar interval (Tazik et al. 1993). Modeling efforts at Fort Hood yielded a prediction that 72 percent of a burned area would be suitable for BCVI occupancy 3 years after a burn (Koloszar and Horne 2000). Results from a large fire at Fort Hood indicated that BCVIs were still increasing in abundance until at least 7 years after the fire (Cimprich 2002). A more recent study at Kerr WMA found that 53.8 percent of winter prescribed burns resulted in increased BCVI use within the same year as the burn, and 92.1 percent of the burns coincided with greater BCVI use within 2 years post fire (Dufault 2004). While cowbird control, deer management, and grazing management may have combined to influence habitat use, Dufault (2004) estimated that 81 percent of the increased use 1 year following a burn could be directly attributed to fire. At 2 and 3 years post-burn, these figures were 78 and 67 percent, respectively. Overall, as noted by Dufault (2004), surveys from Kerr WMA documented an increase in singing males from 27 to 445 during the period 1986 to 2003[2]. This increase was attributed to the prominent influence of prescribed fire combined with brush management, grazing management, white-tailed deer population control, and cowbird removal.

In addition to controlled studies, some wildfires also have provided opportunities for monitoring post-burn occupancy and re-occupancy. The most notable of these opportunities was at Fort Hood, where crown fires burned 4,015 ha (9,917 acres) in February 1996, including 508 ha (1,255 acres) of BCVI habitat (Goering 1998, Hayden et al. 1999). Since then, BCVI abundance has increased on the burned areas while remaining relatively constant on unburned areas of the base (Cimprich 2002). Black-capped vireo abundance increased dramatically between 3 and 4 years after the fire. At 6 years post-burn, there were twice as many point-count detections of BCVIs on burned areas as on unburned areas (Cimprich 2002). Furthermore, BCVIs were detected at a greater percentage of survey points in the burned areas than elsewhere (88 percent vs. 66 percent).

A variety of burn intervals have been suggested for maintaining BCVI habitat, including 4 to 7 years (Campbell 1995), 4 to 10 years (Beardmore et al. 1996), and 25 years (Tazik et al. 1993). Long-term data from Fort Hood and Kerr WMA suggest that the residual influence of fire in creating suitable BCVI habitat may last as long as 20 to 30 years (Tazik et al. 1993, Dufault 2004). However, taking into account the variability in climate and other physical factors across the species range, the actual post-burn use by BCVIs on any one site is likely to be influenced by the season in which the fire occurred, the burning conditions, and the weather patterns after the fire.

---

[2] As of 2005, there were 358 known singing males on Kerr WMA.

Table 3.1. Approximate land area, by Texas county, within each black-capped vireo recovery region; estimated acreage of potential black-capped vireo habitat; and percent of county acreage suitable for BCVI habitat. Table adapted from USFWS (2004), based on roadside survey data from Maresh et al. (1999) and Maresh and Rowell (2000). See text for description of limitations of these data.

Recovery Region/County	**County land area (acres)**	**Potential suitable habitat**	
		(acres)	**(%)**
Region 1			
Bell	611,325	11,004	1.80%
Bosque	632,814	7,594	1.20%
Brown	603,915	36,235	6.00%
Burnet	637,260	11,683	1.80%
Coleman	806,208	20,155	2.50%
Comanche	599,963	10,999	1.80%
Coryell	672,828	4,486	0.70%
Dallas	580,549	900 [1]	0.20%
Erath	695,058	15,060	2.20%
Hamilton	534,508	9,799	1.80%
Hood	269,724	3,147	1.20%
Johnson	466,583	0	0.00%
Lampasas	455,468	4,555	1.00%
Mills	478,686	1,596	0.30%
Montague	590,662	100 [1]	0.20%
Palo Pinto	609,596	11,176	1.80%
Parker	577,980	963	0.20%
Somervell	119,795	1,198	1.00%
Stephens	572,299	7,631	1.30%
Travis	632,814	6,328	1.00%
Williamson	718,276	9,577	1.30%
Region 1 Total	**11,866,311**	**173,186**	**1.47%**
Region 2			
Bandera	506,597	7,599	1.50%
Bexar	797,563	47,854	6.00%
Blanco	454,974	2,275	0.50%
Comal	359,138	3,591	1.00%
Edwards	1,356,030	70,062	5.20%
Gillespie	678,756	58,826	8.70%
Hays	433,732	23,855	5.50%
Kendall	423,852	4,945	1.20%
Kerr	707,655	53,074	7.50%
Kimble	800,033	36,001	4.50%
Kinney	872,157	62,505	7.20%
Llano	597,987	1,993	0.30%
Mason	596,258	35,775	6.00%
McCulloch	684,190	62,717	9.20%
Medina	849,433	62,292	7.30%
Menard	582,920	30,118	5.20%
Real	447,811	31,347	7.00%
San Saba	725,686	6,047	0.80%

Recovery Region/County	**County land area (acres)**	**Potential suitable habitat**	
		(acres)	**(%)**
Schleicher	838,318	1,397	0.20%
Sutton	929,955	46,498	5.00%
Uvalde	995,657	29,870	3.00%
Region 2 Total	**14,638,702**	**678,641**	**4.60%**
Region 3			
Coke	575,016	25,876	4.50%
Concho	634,296	10,572	1.70%
Irion	672,581	0	0.00%
Nolan	583,414	37,922	6.50%
Runnels	672,087	8,961	1.30%
Sterling	590,577	11,812	2.00%
Taylor	585,637	9,761	1.70%
Tom Green	973,674	17,851	1.80%
Region 3 Total	5,287,282	**122,755**	**2.30%**
Region 4			
Brewster	3,961,633	1100 [1]	0.03%
Crockett	1,795,937	125,716	7.00%
Pecos	3,047,486	750 [1]	0.00%
Terrell	1,508,182	2,514	0.20%
Val Verde	2,028,117	344,780	17.00%
Region 4 Total	12,341,355	**473,010**	**3.85%**
TOTAL	**44,133,650**	**1,450,442**	**3.29%**

[1] Habitat area determined from site visits and examination of USGS 1:24,000 topographic maps (Maresh in USFWS 2004).

4.0 Population Status

4.1 Introduction

In this section is an accounting of the known population of BCVI. Ideally, such an assessment would be used to compare against previous range-wide reviews to yield conclusions concerning trends within recovery regions[3] and across the range at large. However, with the notable exception of four relatively well-surveyed areas, there are little data upon which to draw firm conclusions concerning the overall population. We do draw some conclusions concerning the changes in the *known* population of the species; and inasmuch as a large proportion of this known population is concentrated on a few sites, we draw some conclusions concerning the trends on those areas.

The occurrence and abundance data available are primarily for the U.S. breeding range of the species. We do, however, include some recent information on the bird's breeding range in Mexico.

4.2 Approach

The approach we have taken here is to assemble the most recent information available across the BCVI's present distribution. For this, we collected all available federal aid reports from the U.S. Fish and Wildlife Service. We consulted all published and available unpublished records for the species. We sought additional recent data by corresponding with state wildlife biologists, consultants and land managers throughout the species range. To compare current known populations with previous known populations, we draw heavily on information assembled for the Population and Habitat Viability Assessment Report (USFWS 1996) and on the status assessments used in preparing the original listing proposal (Marshall et al. 1985, Grzybowski 1985a).

---

[3] Here we use the most recent recovery regions (or "recovery units") as described in the black-capped vireo population and habitat viability assessment report (USFWS 1996), and these are slightly different from those described in the 1991 recovery plan for the species (USFWS 1991). The 1996 assessment described recovery regions limited to the species' breeding range in Texas, so we additionally refer to the breeding ranges in Oklahoma and Mexico as regions. These are also referred to as Recovery region 1- "North-central Texas", Recovery region 2- "Edwards Plateau", Recovery region 3- "Concho Valley", and Recovery region 4- "Southwest and Trans-Pecos" (USFWS 2004).

With few exceptions, these data are collected and expressed as direct counts of male birds observed during the breeding season, although some workers expressed occurrence as “pairs” or “territories.” For the purpose of the present work, we treat males, pairs and territories as equivalent measures of a breeding unit. Because of inconsistent protocol in species surveys over the years, we used only direct count information where available. Where it appears that a worker estimated a “range” of individual males for an area, we have taken the conservative approach and used only the lowest number reported.

4.3 Known Breeding Populations

At the time of listing, the total known population of BCVIs across the bird’s breeding range was approximately 350 adult birds, including about 191 breeding pairs (Marshall et al. 1985). These numbers comprised 45 to 50 adults, representing about 12 breeding pairs, from four counties in Oklahoma; 280 adults, representing 168 breeding pairs, from 33 sites across 21 counties in Texas; and 24 adults, representing 19 breeding pairs, in Coahuila, Mexico. These counts were based on a combination of records assembled for an earlier status review (Marshall et al. 1985), including surveys in Oklahoma by Grzybowski (1985), and are essentially the figures cited in the proposal that the species be listed as Endangered (Shull 1986). While the BCVI was once considered common in Comanche County, Kansas (Goss 1891) and the original listing included Kansas as part of the historic range (Ratzlaff 1987), there have been no known occurrences of BCVI in Kansas since at least 1956 (Tordoff 1956, Graber 1961).

In 1995, participants in a workshop sponsored by the U.S. Fish and Wildlife Service gathered the known recorded observations of BCVIs throughout the species’ U.S. breeding range from 1990 to 1995. The resulting county-by-county records represent the minimum breeding population known for an area. This effort yielded a total count of 1,803 males–1,636 males from 40 counties in Texas and 170 males from three counties in Oklahoma (USFWS 1996).

For the present status assessment, we gathered a similar dataset of observations recorded from 1996 to 2005, where we used only the most recent data from any one site (i.e., records represent the most recent set of observations, but are not cumulative among years). This dataset is similar in quality to that of the 1996 U.S. Fish and Wildlife Service dataset. For the period 1996 to 2005, the total count of breeding males was 6,269–with 3,515 from 38 counties in Texas, 2,495 from three counties in Oklahoma, and 259 from three states in Mexico (Table 4.1, Fig. 4.1, and Appendix A).[4] When compared to the known occurrences of 1990-1995, BCVIs are now known to occur in nine Texas counties where the species was either not yet confirmed (six counties), was thought to be extirpated (Dallas County), or was not previously known (Callahan and Montague Counties). Likewise, no recent occurrences had been recorded in nine counties where BCVIs had been found during the 1990-1995 period. Of the 33 counties where the BCVI occurred in both time periods, the counts were higher in 19 counties, lower in 10 counties and unchanged in four counties. It is most likely that inconsistent survey efforts throughout much of the species range accounts for inconsistent county occurrence records among the 2 most recent time periods reported here. However, it is noteworthy that most of the counties occupied prior to listing, but from which the species has not been documented since listing, are on the edge of the geographic range (Fig. 4.1). In fact, in the northern-most portion of the breeding range, the species is currently known in only 4 of the 24 counties from which it was previously known.

The species' breeding range in Mexico has been only sparsely surveyed. At present, the entire inventory of 259 males from three states in Mexico can be attributed to some limited survey efforts during the past 3 years (Table 4.1, Appendix A). While the counts are relatively low, the population densities indicated by surveys in Mexico are relatively high and appear to hold promise for revealing major population centers for the species. For example, Benson and Benson (1990) documented 28 singing males at four sites in the Sierra del Carmen mountain range and estimated a breeding population of 6,301 ± 3,162 pairs ($P<0.1$) for the region, based on an extrapolation of their density estimates (1.43 pairs per km^2). This estimate was significantly higher than the 48 to 131 pairs estimated by Marshall et al. (1985). Scott and Garton (1991) called into question the methodology used in the original population estimate for northern Mexico, and Benson and Benson (1991) subsequently revised their techniques and produced a new estimated population size using a distance algorithm (Burnham et al. 1980). Upon reanalysis, Benson and Benson (1991) determined BCVI densities of 1.65 singing males per km^2

---

[4] A similar comparison was recently prepared by Maresh (2005), and we used that document and many of the same sources, to assemble the records in Appendix A, resulting in the summary of Table 4.1. We updated several of the records and added the results of surveys conducted on private lands in Texas. Another substantial difference in the final figures is the result of some extrapolated population numbers used by Maresh (2005). Our approach was to use (as much as we could tell) only known and documented occurrences.

for northern Mexico, which they extrapolated to a minimum of 3,395 singing males (7,286 ± 3,891).

McKinney (1998) corroborated high density estimates in northern Coahuila, finding 26 singing males in a 4-ha area and another 20 singing males in a 6-ha area. McKinney's results suggest a population density in this part of Mexico much greater than that known within the Texas and Oklahoma breeding ranges, where males typically defend breeding territories of 1 or 2 ha (mean=1.5 ha; Graber 1961) to 10 ha (mean=3.6 ha; Tazik 1991) in size. However these figures were based on a small number of sites. In northern Coahuila, Farquhar and Gonzalez (2005) estimated breeding densities of 3.29 singing males per ha (± 0.37), which is three to six times as large as the typical densities found in Texas and Oklahoma.

4.4 Abundance Patterns

At the time of listing, it was thought that the largest concentrations of BCVIs were in the immediate vicinity of Austin, Texas (Shull 1986, Ratzlaff 1987). This was predicated on the work of J. T. Marshall and R. B. Clapp, who found 33 pairs of BCVIs at Travis County's 227-acre (~100-ha) Wild Basin Wilderness Preserve, and on the approximately 20 males documented by C. Sexton and others in areas just west of Austin (Marshall et al. 1985). Other significant concentrations known at the time of listing included 34 pairs at Kerr WMA in Kerr County, Texas and 15 to 17 males and 3 females at Wichita Mountains WR in Comanche County, Oklahoma.

While the known breeding population today is at least 30 times greater than what was documented at the time of listing, these count data do not clearly establish that the overall population itself has increased by such a margin. Most of the known occurrences of BCVIs are concentrated on a small number of properties. In fact, about 75 percent of the known breeding population is found on four properties–Fort Hood Military Reservation (TX), Kerr WMA (TX), Wichita Mountains WR (OK), and Fort Sill Military Reservation (OK)–two of which (the Oklahoma properties) are adjacent. The other 25 percent of known occurrences are from at least 52 other properties distributed throughout the species' range. Many of these occurrences are on private lands, which account for more than 80 percent of the land within the species' geographic range. Where private lands are accessible, and have been included in systematic surveys, the species is often found. For example, on private lands in Texas, Magness (2003) documented 11 males on seven sites in Bandera and Real Counties and Juarez (2004) documented male BCVIs at 26 sites on private lands in Coryell and Hamilton Counties. Surveys on private lands in other counties also have yielded BCVI occurrences (Appendix A). Despite the increased survey efforts since 1995, most of the BCVI breeding range in the U.S. is on private lands that are as yet unsurveyed.

From 1995 to the present, the known breeding population at three of the four major population centers increased substantially (see Appendix A). At Wichita Mountains WR and Fort Sill (combined), the number of territorial males documented increased from 150 to 2,474; at Fort Hood, the count increased from an estimated 300 to 1,847 in 2003[5]. However, in Kerr County, Texas (Kerr WMA and other private lands), the known population decreased from 602 in 1995 to 436 in 2005.

Researchers at Fort Sill and Wichita Mountains WR have reliably documented population expansions since listing (Grzybowski 2005). Results of systematic surveys of fixed areas at Fort Hood have demonstrated substantial increases in numbers of territorial males since 1987 (Kostecke et al. 2005). At Kerr WMA, intensive habitat management has contributed to an increase in known territorial males from 27 in 1986 to 445 in 2003 (e.g., Dufault 2004).

While it remains unclear as to whether the species has increased or decreased in abundance over the time period represented by these surveys, it appears likely that the species has increased in abundance at Kerr WMA, Wichita Mountains WR, Fort Sill and Fort Hood.

4.5 Population Genetics

For declining and isolated populations, maintenance of genetic variation is a serious concern because fragmented populations often lose their genetic diversity over time. Genetic variation is important in providing flexibility in response to changing environments (Allendorf and Leary 1986, Hedrick and Miller 1993) and depletion of genetic variation within a population is common in small populations due to random drift and founder events (Wright 1931, Allendorf 1983, Lande and Barrowclough 1987). At particularly low population levels, genetic loss may

---

[5] The most recent surveys at Fort Hood suggest a population in the range of 4,834 to 8,261 (95% Confidence Interval) males (Cimprich 2005) – these figures were based on density estimates derived from distance sampling methods in representative habitats. We used the more conservative numbers for Appendix A based on direct observations rather than the extrapolated estimates.

be accelerated through inbreeding (Gilpin and Soule 1986). Fragmentation of once-continuous habitat can result in a loss of genetic variation and an increase in population differentiation (Stangel et al. 1992), leading many species to experience population declines.

The BCVI Recovery Plan (USFWS 1991) recognized concerns about genetics as an important element of population viability analysis (Gilpin and Soule 1986, USFWS 1996). Such concern might be warranted considering the large distances that separate many of the known populations (i.e., Oklahoma and Mexico) from the Texas populations. In fact, the distances between some BCVI populations are greater than those observed among some subspecies of vireos (Avise et al. 1982, Johnson et al. 1988, Johnson 1995). Large geographic distances between BCVI populations may be a concern because there is no evidence to suggest that dispersal is occurring between such isolated populations. Most (96%) adult male (>1 year old) BCVI return to breed on the same territory as the previous year (Grzybowski 1995). Many females and yearling males disperse to new sites both within and between seasons, but the longest known movement was 10 km (Grzybowski 1995). This suggests that dispersal over very large distances between populations is most likely rare, although Grzybowksi (1995) suggested that juvenile dispersal may be considerably underestimated.

Recent efforts to characterize heterozygosity and population structuring in the BCVI found surprising variability, considering the apparent limited dispersal between isolated populations, within four geographically isolated populations (Wichita Mountains WR, Fort Hood Military Reservation, Kerr Wildlife Management Area, and Kickapoo Caverns State Natural Area) in Texas and Oklahoma (n= 72) (Fazio 1994, Fazio et al. 2004). Within-population heterozygosity was high (0.058) relative to other species within Vireonidae (0.023-0.056, Johnson et al. 1988) and comparable to the mean for birds (0.068, Ward et al. 1992). The highest heterozygosity was within the Wichita Mountains population (0.067), which is surprising considering its relative isolation and historically small size. Generally, for subpopulations that remain small over periods of 50 or more generations, substantial loss of heterozygosity is predicted (Allendorf 1986, Lacy 1987), but that does not appear to be case in this situation.

It has been suggested that the relatively high levels of heterozygosity and the BCVI association with successional habitats may suggest source-sink population dynamics and a metapopulation structure (Hanski and Gilpin 1991), with extinctions and recolonizations occurring in satellite groups within the Wichita Mountains (e.g., Fort Sill and Wichita Mountains WR) (Fazio et al. 2004).

There was significant differentiation between BCVI populations (mean $\theta=0.17$; bootstrap 95% confidence interval 0.004-0.35), as one would expect for disjunct populations, but gene flow between populations appeared sufficient to maintain substantial within-group variation, perhaps due to dispersal by juveniles (Fazio et al. 2004). The authors found that patterns of gene flow suggested a long-term pattern of limited gene flow, at levels less than those observed for other migratory species (Rockwell and Barrowclough 1987). They concluded, nevertheless, that gene flow was sufficient among populations to overcome the effects of genetic drift (Slatkin 1985). It is unclear what patterns of gene flow exist within and among BCVI populations in southwestern Texas or in northern Mexico, as no genetics studies have been conducted at this time on those populations.

Greater genetic similarity was found between the Kickapoo Caverns and Kerr populations and between the Wichita Mountains and Fort Hood populations (Fazio et al. 2004). The authors suggested that if Wichita Mountains is acting as a sink, then it is likely that colonists may have come from populations at Fort Hood.

Table 4.1. Number of known black-capped vireos, based on the most recent surveys, within each of the recovery regions and in other areas of known occurrence. See Appendix A for detailed occurrence records.

Region	Known breeding population (males)
Texas Recovery region 1	2,110
Texas Recovery region 2	1,018
Texas Recovery region 3	149
Texas Recovery region 4	236
Texas (other)	2
Oklahoma	2,495
Mexico	259
Total	6,269

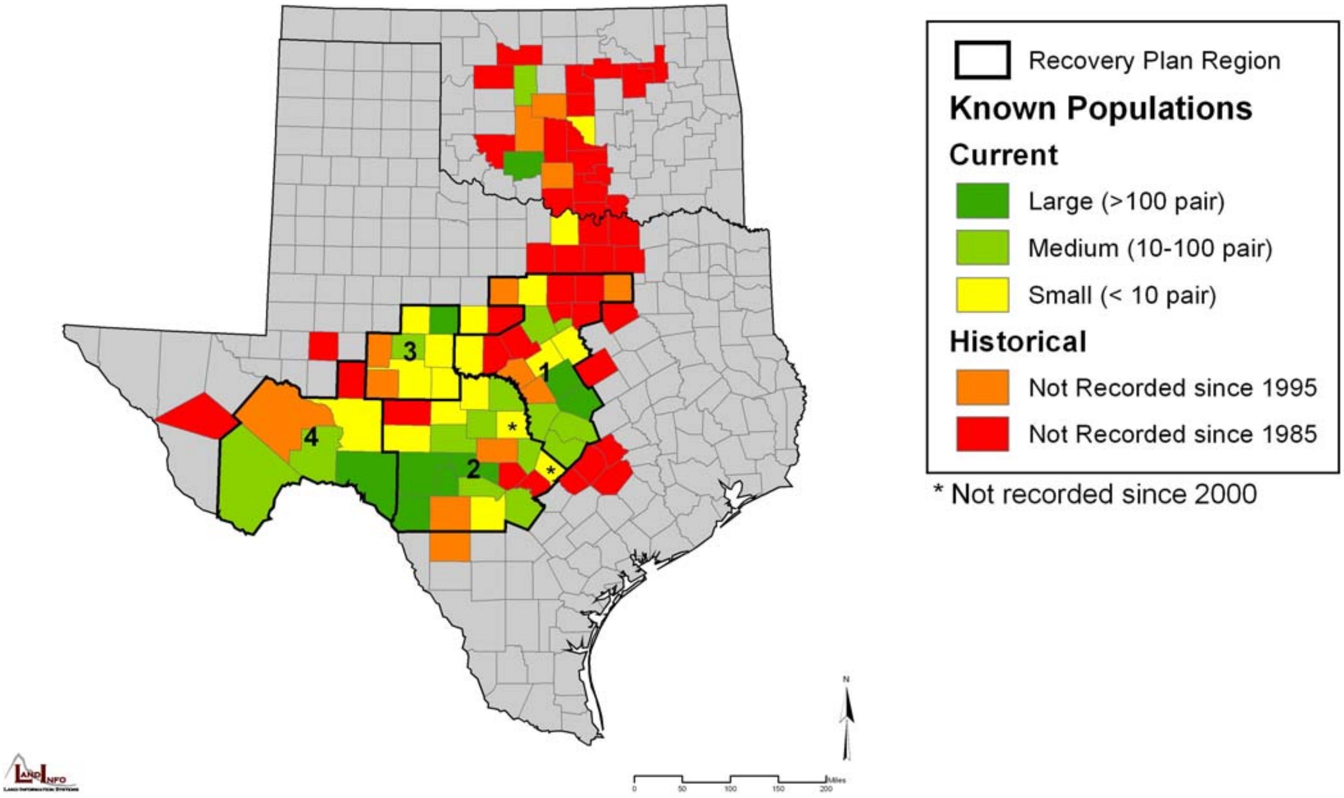

Figure 4.1. Current relative population sizes for known occurrences of black-capped vireos throughout the contiguous U.S. breeding range of the species, including areas where the species has been known to occur but has not been documented in the past 10 and 20 years, respectively. Relative population sizes by county are based on figures available in Appendix A. Note: the species breeding range in 3 states of Mexico are not included in this figure due to limited data from those states.

5.0 Analysis of Threats

5.1 Introduction

At time of listing, the major threats to the BCVI were identified as habitat loss through land use conversion, vegetation succession, grazing and browsing by domestic and wild herbivores, and brood parasitism by brown-headed cowbirds (Ratzlaff 1987). Using the resources identified by our review team and review panel, we have collected and reviewed the available information concerning these threats and attempted to address the ways they have changed since listing. In many cases, we depend on summary analyses of secondary data that provide an indirect assessment of the threat.

5.2 Habitat Conversion and Land Use Change

Habitat loss (through conversion, fragmentation and land use change) was a threat factor considered in the listing of the BCVI. At the time of listing, much of the concern focused on impending urban development in the greater Austin, Texas area (Shull 1986), where a large portion of the known population of the species was found.

Because there are no direct measures of the amounts and distribution of suitable habitat for the species, it is not possible to conduct a direct trend analysis for this threat factor. Other available data do not clearly indicate the rates of habitat loss from land use conversion. However, data on land use change and land ownership size distribution may be helpful in understanding some of the trends that might affect the species.

5.2.1 Land use

The proposed current U.S. breeding range of BCVI spreads across 98 counties in Texas and Oklahoma. This area comprises 68.8 million acres, about 80 percent of which was classified as farm and ranchland in 2002 (Table 5.1, USDA Agricultural Statistics Service). In the statistical accounting of the Agricultural Census, essentially all suitable habitat for BCVIs is likely classified as “rangeland.” According to the 2002 Agricultural Census, approximately 33.9 million acres of the U.S. breeding range was classified as rangeland (Table 5.1). In the Oklahoma portion of the breeding range, the area counted as farm and ranchland and that portion considered rangeland have remained relatively stable since 1992–in fact, the statistics suggest a slight increase in both over this period. In contrast, across the Texas portion of the breeding range, comparisons of the 1992 and 2002 Agricultural Census figures show a net loss of approximately 2.3 million acres of farm and ranchland and a net loss of 3.2 million acres of rangeland. This amounts to a 4.9 percent loss of farm and ranchland and an 8.6 percent loss of rangeland. This suggests that rangeland was lost not only because it was converted to uses other than farming and ranching, but also because it was converted to farm and ranchland (such as cropland or non-native pastures). This trend has reduced the overall area that would otherwise have been available for the development of suitable habitat across portions of the species' range in Texas. However, the amount of suitable breeding habitat actually lost to land use conversion is unknown.

5.2.2 Ownership fragmentation

As was the case with land use conversion, the trends in ownership fragmentation across the species' range in Oklahoma were quite different than those in Texas. In Texas, large ownerships (more than 2,000 acres) declined from 31.2 million acres in 1992 to 28.4 million acres in 2002–a net loss of 9.1 percent. In Oklahoma, where smaller ownerships were already prevalent across the region, there was actually an increase in larger ownerships (from 2.9 million acres to 3.3 million acres) over the same period (2002 USDA Census of Agriculture).

The total number of farm and ranch ownerships of less than 500 acres increased dramatically across the species' range in both states during the 1992-2002 period. In Texas, a 40 percent increase in such ownerships resulted in a net gain of 19,571 new farms and ranches. In Oklahoma, a 39 percent increase resulted in 6,026 new ownerships. Oklahoma's gain in the number of smaller farms and ranches was the product of fragmented mid-size ownerships (500 to 2,000 acres).

Although the relationship between ownership size distribution and wildlife habitat fragmentation is not entirely known, there is some evidence that the fragmentation of large farm and ranch ownerships (more than 2,000 acres) into smaller parcels may change land use and habitat patterns unfavorably for many species of wildlife (Wilkins et al. 2003, Kjelland et al. 2006). However, there are no studies directly implicating ownership fragmentation or its outcome as a conservation threat to BCVIs. While habitat loss from land use conversion can legitimately be considered a direct threat, the continued fragmentation of large land parcels might be considered an indirect threat simply because of the consequences associated with an increase in small ownerships. While the fragmentation and loss of larger ownerships might pose a challenge for maintaining large, intact areas of suitable habitat, new management styles on smaller ownerships may create suitable habitat for BCVIs. Many ranches in Texas have traditionally been managed for wildlife conservation, and BCVIs are now known to occur on several ranches. Some newer landowners also seem interested in management systems that could result in the development of suitable habitats for BCVIs (Sanders 2005).

Table 5.1 Area in farm and ranchland, and rangeland area, by state and BCVI recovery region, 2002 and change since 1992.

State[1]	**Recovery region**	**Farm and ranchland**			**Rangeland**		
		2002	Change since 1992		2002	Change since 1992	
		Area (acres)	Area (acres)	Percent	Area (acres)	Area (acres)	Percent
Texas	1. North-central Texas	8,801,998	-208,221	-2.3	5,147,140	-385,632	-7.0
	2. Edwards Plateau	12,142,910	-458,496	-3.6	9,934,658	-590,079	-5.6
	3. Concho Valley	4,641,999	-582,747	-11.2	3,408,370	-674,325	-16.5
	4. Southwest and Trans-Pecos	9,937,848	-1,176,238	-10.6	9,728,496	-1,217,865	-11.1
	Not in recovery region	9,720,610	106,116	1.1	5,691,449	-302,873	-5.1
Texas total		45,245,365	-2,319,586	-4.9	33,910,113	-3,170,774	-8.6
Oklahoma total		9,673,720	359,915	3.9	4,628,535	198,803	4.5
TOTAL		54,919,085	-1,959,671	-3.4	38,538,648	-2,971,971	-7.2

[1] Includes only those counties in the proposed current U.S. breeding range of the black-capped vireo.

5.3 Vegetation Change

"Vegetational succession" was listed as a major threat to the BCVI in the original listing document for the species (Ratzlaff 1987). In general terms, this vegetational succession is better described as an increase in the canopy cover and stature of woody vegetation. Black-capped vireos breed in shrubland and scrub habitat that, in many parts of the species' range, is considered mid- to early successional. Habitats within the BCVI range have changed substantially in the past century. Perhaps the most apparent change is the increased abundance of woody plants (Fuhlendorf and Smeins 1997). Increased stature and cover by woody vegetation often results in loss of breeding habitat suitability (USFWS 1991, Grzybowski 1995).

Here, we describe the threat posed by woody plant encroachment to BCVI habitat across a major portion of the species' breeding range in Texas and Oklahoma. We discuss various factors that contribute to such vegetational changes. Finally, to establish an index for determining the influence of vegetational changes on the BCVI, we examine abundance trends for a group of birds that prefer increased shrub and tree cover and discuss the implications of these findings.

5.3.1 Invasive woody plants

The increase in shrub and tree cover across arid and semi-arid rangelands in Texas and Oklahoma is well documented (Bogusch 1952, McPherson et al. 1988, Smeins and Merrill 1988, Archer 1990, McPherson and Wright 1990, Scanlan and Archer 1991, Snook 1985). Much of the increase can be attributed to the expansion of junipers (*Juniperus* spp.) beyond their historic range (Foster 1917, Tharp 1926, Fowler and Dunlap 1986, Ansley et al. 1995, Engle et al. 1995, Thurow and Thurow 1997, Thurow et al. 1997, Ueckert 1997). This encroachment by junipers corresponds with a period of more intensive livestock grazing and the suppression of fire (Archer 1994, Fuhlendorf et al. 1996, Smeins et al. 1997).

The Natural Resource Conservation Service's (NRCS) Texas State Technical Committee recognized that Ashe and redberry juniper (*J. smallii*) and (in some cases) honey mesquite (*Prosopis glandulosa*) had the potential to reduce the quality of BCVI habitats (USFWS 2004). According to some generalized mapping efforts, Ashe juniper is distributed across a minimum of approximately 18 million acres (Fig. 5.1), while redberry juniper (Fig. 5.2) and honey mesquite

(Fig. 5.3) are distributed across 10 and 58 million acres, respectively (USFWS 2004). All three of these species are native but considered invasive.

Ashe juniper is found mostly in the Edwards Plateau, and it is the invasive species with the most effect on BCVI habitats in the eastern two-thirds of the species' U.S. breeding range. Ashe juniper is most common in the eastern and southern portions of the Edwards Plateau, while redberry juniper is found in the northern and western portions (Lyons et al. 1998). The distribution of these two juniper species overlaps along the western margin of the BCVI's breeding range, and in some of those areas redberry juniper is the primary invasive species.

In the absence of fire, grazing or similar disturbance, junipers can out-compete native grasses and change the structural characteristics of native rangelands (Arend 1950, Archer 1994, Ansley et al. 1995). In the higher rainfall areas of the eastern Edwards Plateau and across parts of the Rolling Plains, invading juniper can develop into dense stands that are generally unsuitable for BCVIs (Keddy-Hector 1992). In the drier portions of the western Edwards Plateau, however, many areas have remained relatively free of juniper and, where there are other preferred brush species, BCVIs remain unaffected by juniper invasion (Keddy-Hector 1992).

Fire can help control both Ashe and redberry juniper. Ashe juniper does not sprout after disturbance and is easily killed by fire if the entire shrub or tree is consumed (Fonteyn et al. 1988), while redberry juniper does re-sprout from the roots. Thus, suppressing fire contributes to invasion by these species (Fuhlendorf et al. 1996, U.S. Fish and Wildlife Service 1996, Lyons et al. 1998). Overgrazing combined with extended drought is also a factor in juniper invasion (Lyons et al. 1998, McPherson et al. 1988). Both redberry and Ashe juniper are phreatophytes, meaning that they can develop deep root systems and exist on water from a permanent ground supply or the water table. This allows them to out-compete other woody species during times of drought.

Honey mesquite has also increased on much of the south Texas brushlands, the Rolling Plains and other semi-arid rangelands (USFWS 2004), making these areas less suitable for many grassland bird species (Magness 2003). Honey mesquite is native to parts of Texas and has increased in abundance and density within its historic range, rather than expanding its geographic range (Tharp 1926, Bogusch 1952, Johnston 1963). Honey mesquite may affect BCVI habitat in some portions of the breeding range, but it has not received (nor deserved) as much attention as juniper.

In Oklahoma, eastern redcedar (*J. virginiana*) is widespread and has degraded BCVI habitat across much of its range (Graber 1957, Penfound 1968, Hayden and Tazik 1991, Greenman 1995), with the possible exception of the most southwestern and panhandle counties (Crockett 1985). The acreage of eastern redcedar in Oklahoma was estimated to have increased by 141 percent between 1950 and 1985, to a total of 3.54 million acres (Snook 1985).

As is the case with juniper in Texas, the suppression of fire in Oklahoma is a primary contributor to invasion by eastern redcedar (U.S. Fish and Wildlife Service 1996). Without fire, eastern redcedar invades aggressively (Arend 1950, Blan 1970, Blewett 1986, Stritzke and Bidwell 1990); after burns, there are far fewer of the plants (Penfound 1969, Rollins 1985, Greenman 1995). Much of the BCVI's former range in west-central Oklahoma is now substantially covered in eastern redcedar, as are large areas of north-central and central Texas (Grzybowski 1995). Modelling studies have estimated that at the current rate of expansion in Oklahoma, juniper species will overrun substantial areas of remnant grassland over the next 10 years (Coppedge et al. 2004).

5.3.2 Influence of fire

Fire can help maintain suitable successional stages of vegetation for the development of breeding habitats (Ratzlaff 1987). As a consequence, BCVIs are often found in areas where fires have recently occurred, and the highest BCVI concentrations typically occur in areas recovering from hot fires (Graber 1957, Marshall et al. 1985, Grzybowski et al. 1994). With the proper burning conditions, fires can kill or retard invading junipers and favor the regrowth of fire-adapted oak and sumac species. This produces areas of dense foliage at the low level required by BCVIs (USFWS 1991, Campbell 1995, Grzybowski 1995).

Throughout much of the species' U.S. range, the appropriate successional stages for nesting habitat were historically maintained by fire. Therefore, fire suppression is considered to be an indirect cause of habitat loss (Grzybowski 1995, Gehlbach 1988, Smeins and Merrill 1988). Fire suppression, in combination with heavy grazing and browsing, can transform a mixed-oak savanna into oak woodland with dense under-story and mid-story juniper (Fonteyn et al. 1988), making it unsuitable as nesting habitat for BCVIs (Marshall et al. 1985). Although this dynamic is important in the eastern portion of the species' U.S. breeding range, there are portions of the breeding range in southwestern Texas and in Mexico where disturbance by fire is substantially less important in maintaining suitable habitat structure (Farquhar and Gonzalez 2005).

Several studies have addressed the beneficial effects of fire on BCVI habitat and populations. Black-capped vireos fully recolonized burned areas by the second year following a burn at Wichita Mountains WR (Grzybowski 1989, Grzybowski 1990a); results from Fort Hood, Texas suggested a similar interval (Tazik et al. 1993). A more recent study at Kerr WMA found that 53.8 percent of winter-season prescribed burns resulted in increased BCVI numbers within the same year as the burn, and that 92.1 percent of the burns produced larger numbers of BCVIs within 2 years of the burn (Dufault 2004).

Some wildfires have also provided opportunities for monitoring post-burn re-occupancy. One example occurred at Fort Hood, where crown fires burned 4,015 ha (9,917 acres) in February 1996, including 508 ha (1,255 acres) of BCVI habitat (Goering 1998, Hayden et al. 1999). On areas that were burned, BCVI numbers have increased with time, while remaining relatively constant on unburned areas of the base (Cimprich 2002). The unequivocal and substantial effect of fire on BCVI breeding habitat cannot be ignored. The social, legal and political constraints to using prescribed burning as a management tool are, in fact, a concern for the species.

5.3.3 Trends in woodland birds as an index of vegetation change.

Birds that depend on grassland and savanna habitats appear to be declining more than any other North American avian group (Askins 1993, Peterjohn and Sauer 1999). Across the southern Great Plains, the invasion of woody plants has been implicated as a significant factor in this trend (Brennan and Kuvlesky 2005). A general increase in woodland habitats could result in a region-wide shift in breeding bird assemblages with a greater representation by those species that prefer wooded habitats. While the BCVI is not a grassland obligate per se, its preference for short-stature shrubland and scrub means that a general increase in woodland habitats would likely result in loss of suitable breeding habitat. To gain an index for such a change within the specific U.S. breeding range of the BCVI, we analyzed trends for bird species known to commonly use shrubland and woodland habitats. The species selected were the white-eyed vireo (*V. griseus*), blue-gray gnatcatcher (*Polioptila caerulea*), Bell's vireo (*V. bellii*), painted bunting

(*Passerina ciris)*, and yellow-breasted chat (*Icteria virens*)[6]. These species characteristically inhabit denser stands of scrub and thickets than do BCVIs (Ehrlich et al. 1988).

While these species commonly use habitat that is also used by BCVI, they also use habitats with woody vegetation developed beyond the point of suitability for BCVI breeding. For instance, white-eyed vireos tend to select habitats in large woodland patches (Rodewald and Vitz 2005) and with dense shrubs (Annand and Thompson 1997). At study sites in Hamilton and Coryell Counties, Texas, the sites occupied by white-eyed vireos had significantly ($P \leq 0.05$) more woody vegetation than unoccupied sites (Juarez 2004). In Bandera, Real, Kerr, Uvalde and Medina Counties, Texas, sites occupied by white-eyed vireos also had significantly ($P<0.05$) more juniper cover, at both the local and landscape scales, than unoccupied sites (Magness 2003). In the same counties, sites occupied by blue-gray gnatcatchers also had significantly ($P<0.05$) more juniper cover at the landscape scale than unoccupied sites (Magness 2003). Other research in juniper-dominated landscapes has shown that breeding blue-gray gnatcatchers select habitats with the most shrub cover (Pavlacky and Anderson 2001). Bell's vireos prefer habitat with dense shrub layers (Goldwasser 1981, Franzreb 1989b, Brown 1993). At sites across the Concho Valley (in Reagan, Tom Green, Irion and Schleicher Counties, Texas), habitat occupied by painted buntings had significantly ($P<0.05$) more juniper and mesquite cover than unoccupied sites (Magness 2003). In Oklahoma, painted buntings were generally more abundant where there was more closed forest than open country (Brennan and Schnell 2005), and it has been predicted that painted buntings will increase in abundance in Oklahoma with continued juniper encroachment (Coppedge et al. 2004). Yellow-breasted chats are often more abundant, and are more likely to nest, in shrubby patches that are conspicuously large and dense (Annand and Thompson 1997, Burhans and Thompson 1999, Woodward et al. 2001, Rodewald and Vitz 2005).

We used data from the USGS North American Breeding Bird Survey (BBS). The BBS data is useful for examining geographic patterns of abundance and population change over the breeding range of a species or suite of species (Sauer and Droege 1992, Maurer and Villard

---

[6] Species were chosen in consultation with the review panel to represent species known to use habitats of greater woodland cover than that generally used by BCVIs.

1994, James et al. 1996).[7] There are currently 217 BBS routes in Texas and 70 routes in Oklahoma (Fig. 5.4)[8]. We confined analysis to only those routes within the recent breeding range of BCVIs in Texas and Oklahoma (Fig 5.4) and used only the best quality data available (i.e., "Type 1," as designated by the U.S. Geological Survey Patuxent Wildlife Research Center). We also followed route selection criteria, as described by James et al. (1996), considering the data only from those routes that maintained a fairly consistent level of operation, and omitting routes that were run sporadically, were discontinued, or were run consistently only recently. Thus, for displaying trends, we selected routes only if they 1) had been surveyed eight or more times and 2) had been surveyed at least once in each of the four periods 1966-1975, 1976-1987, 1988-1994 and 1995-2004. We also used paired t-tests to make comparisons between periods (James et al. 1986).[9] Thus, the division of data into these specific periods was intended to provide a primary comparison between the 10-year period prior to BCVI listing and the most recent 10-year period for which data is available. For use in these paired comparisons, we only considered routes that were in operation for at least 5 years during any of the periods being compared.

While the overall population trends for these five species are variable (Figs. 5.5 to 5.9), they do tend to have a single common feature – all five species appear to have increased in relative abundance in the 10-year period ending in 2004. For the white-eyed vireo (Fig. 5.5), the average observations per route for the 10-year period ending 2004 was approximately 4-fold higher than that for any other prior period ($P<0.01$). For the blue-gray gnatcatcher (Fig 5.6), relative abundance increased in the 10-year period ending 2004 when compared to both 10-year periods before to the 1987 listing of black-capped vireos ($P<0.05$). For the yellow-breasted chat, while mean observations per route remained relatively low, the 10-year average for the period ending 2004 was higher than that of the prior seven years (1988-1995, $P<0.10$) but was

---

[7] BBS data are collected by observers along pre-defined 40-km routes. Observers stop every 0.8 km to record all birds seen or heard during a 3-minute period. Not all routes are regularly surveyed, and these data often have gaps because of poor weather, improper timing of survey, etc. Nevertheless, for the purposes of examining trends across space and time, BBS data can be useful.

[8] There are no data for these species in the breeding range of the BCVI in Mexico.

[9] The data represent number of observations per route for each species during each year that a route was surveyed. For statistical comparisons, these observations were square-root transformed to stabilize variance per James et al. (1986).

statistically indifferent from that observed in previous periods. For Bell's vireo (Fig. 5.8) and painted buntings (Fig. 5.9), the pattern suggests a decrease during the seven-years immediately following the BCVI's listing ($P<0.05$), followed by an increase in relative abundance during the final 10-year period ($P<0.05$).

It is also notable that cowbird abundance is known to decrease significantly with increasing forest cover and increase with increasing edge density (meters per hectare) (Donovan et al. 2000, Thompson et al. 2000). Magness (2003) found, at the local and landscape scales, that sites brown-headed cowbirds occupied had significantly ($P<0.05$) less juniper cover than unoccupied sites in Bandera, Real, Kerr, Uvalde and Medina Counties, Texas. Similarly, Juarez (2004) found that the sites cowbirds occupied had significantly ($P\leq0.01$) less woody vegetation and significantly ($P\leq0.01$) less juniper cover than unoccupied sites in Hamilton and Coryell Counties, Texas. The declining number of cowbirds across the BCVI range (see Section 5.5 on Brood Parasitism) further suggests that vegetation changes may be influencing avifauna at a regional scale – and the implications for BCVI might be confounded by the impacts.

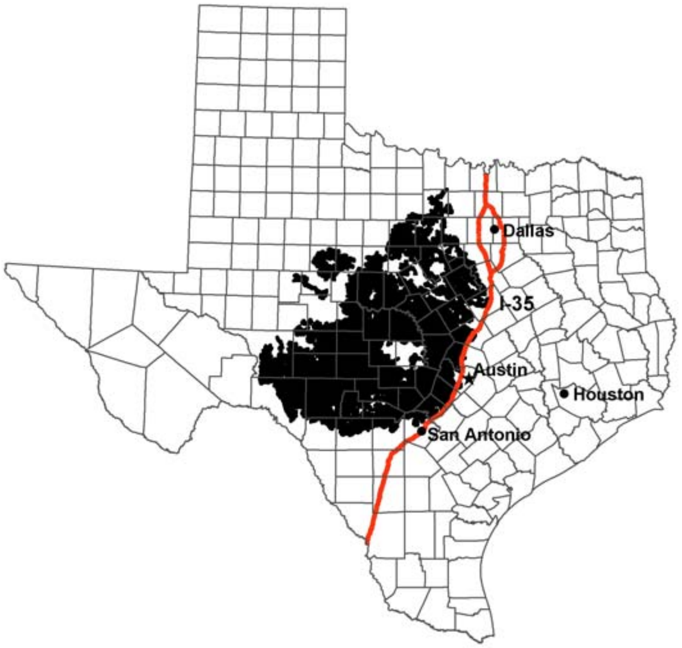

Figure 5.1. Distribution of Ashe juniper in Texas.

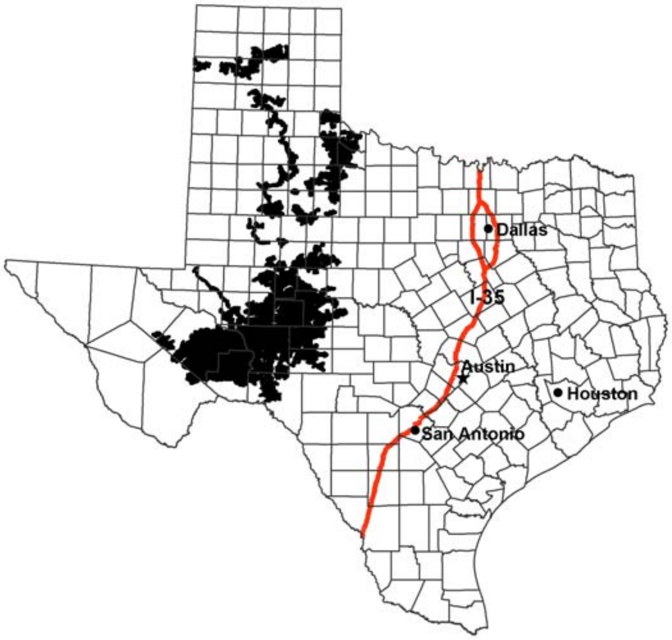

Figure 5.2. Distribution of redberry juniper in Texas.

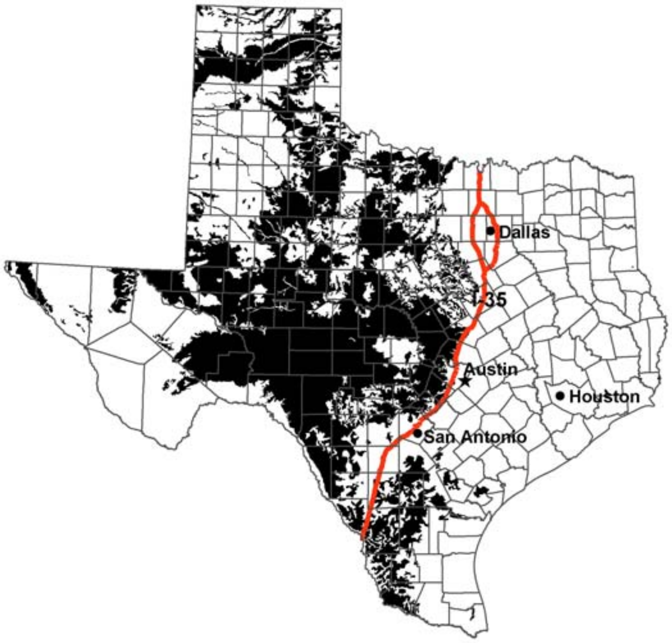

Figure 5.3. Distribution of honey mesquite in Texas.

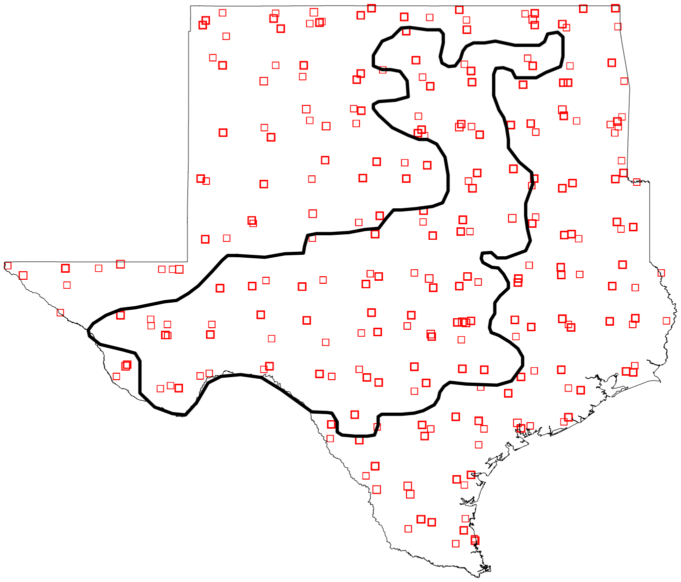

Figure 5.4. Approximate center-points for 287 Breeding Bird Survey routes across Texas and Oklahoma. The thicker black line represents the current estimated U.S. breeding range of black-capped vireos.

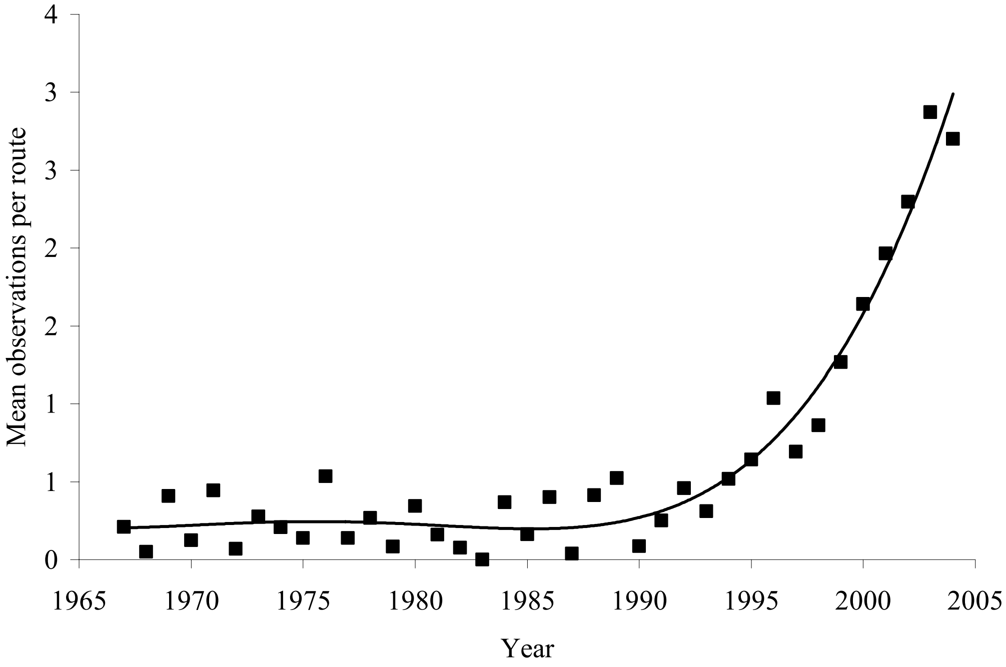

Figure 5.5. Relative abundance for the white-eyed vireo in the breeding range of the black-capped vireo in Texas and Oklahoma, 1967-2004. Data Source: Sauer et al. (2005).

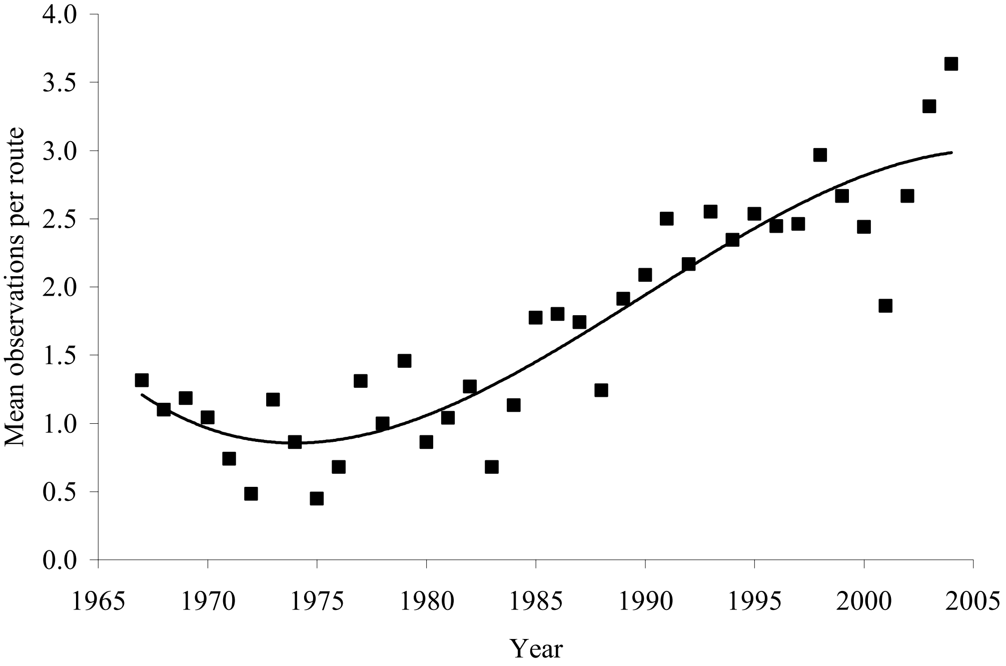

Figure 5.6. Relative abundance for the blue-gray gnatcatcher in the breeding range of the black-capped vireo in Texas and Oklahoma, 1967-2004. Data Source: Sauer et al. (2005).

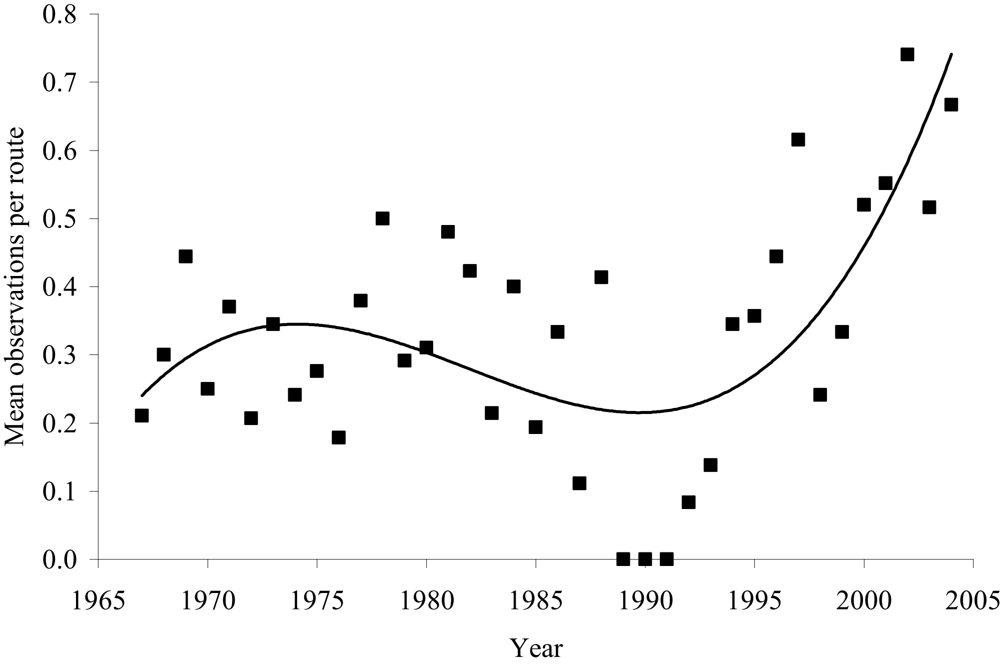

Figure 5.7. Relative abundance for the yellow-breasted chat in the breeding range of the black-capped vireo in Texas and Oklahoma, 1967-2004. Data Source: Sauer et al. (2005).

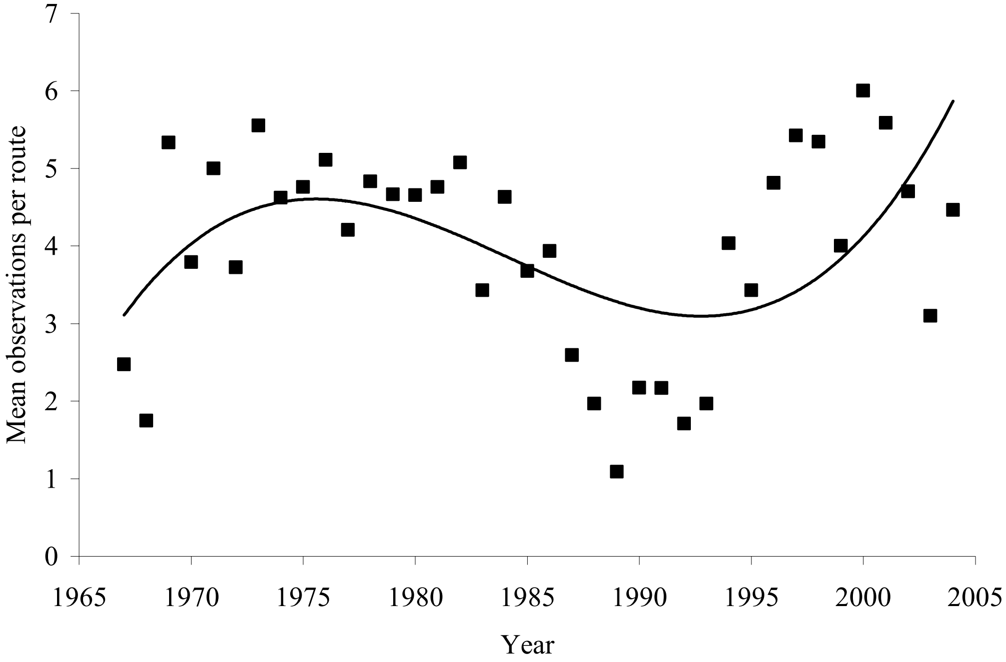

Figure 5.8. Relative abundance for Bell's vireo in the breeding range of the black-capped vireo in Texas and Oklahoma, 1967-2004. Data Source: Sauer et al. (2005).

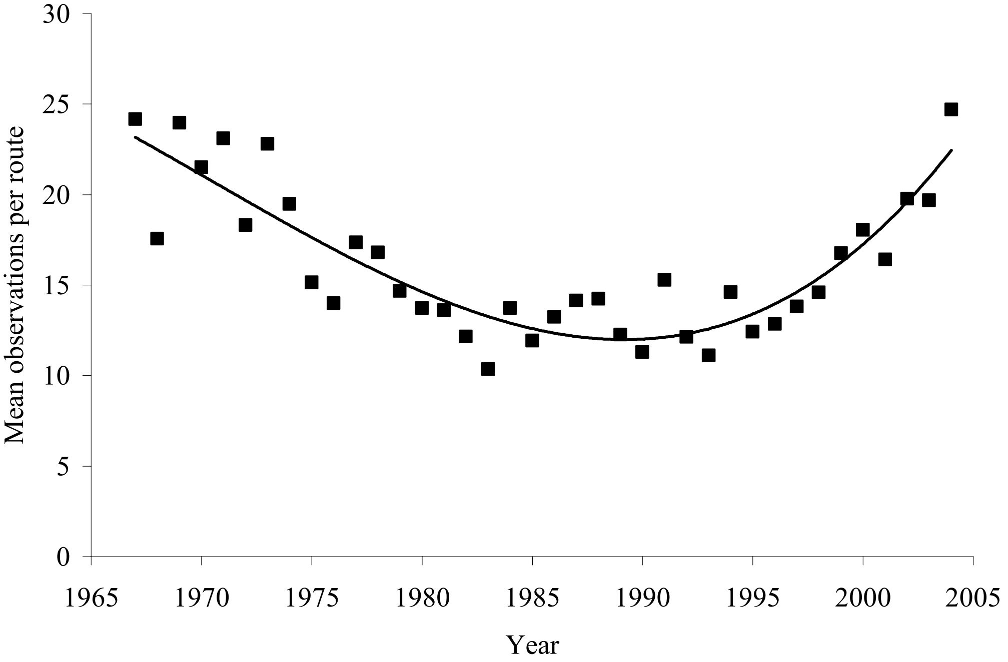

Figure 5.9. Relative abundance for the painted bunting in the breeding range of the black-capped vireo in Texas and Oklahoma, 1967-2004. Data Source: Sauer et al. (2005).

5.4 Grazing and Browsing

The grazing of cattle, sheep, goats and other herbivores was said to be one of the primary threats to the BCVI in the original listing of the species as endangered (Ratzlaff 1987). Overgrazing, particularly by browsers, typically removes vegetation at the heights needed by BCVIs (Grzybowski 1995). Before the listing it was determined that intensive grazing, primarily by goats and sheep in many areas of the Edwards Plateau of Texas, had reduced the habitat available to BCVIs (Grzybowski 1995). The association between brown-headed cowbirds and concentrations of cattle (Lowther 1993, Ortega 1998) may have compounded the threat of grazing to BCVIs.

Here, we provide an overview of the threat of overgrazing to BCVIs across their breeding range. We present background information on the effects of grazing on vegetation communities and passerine birds, discuss known effects on BCVIs in particular, then examine trends in cattle and goat densities, as well as deer numbers, across the range of the BCVI and discuss the implications of these findings.

5.4.1 Grazing on native rangelands

Grazing by domestic livestock is the principal economic use of native rangelands in the western United States (Platts 1991, Lauenroth et al. 1994). Throughout the BCVI's breeding range in Texas, Oklahoma and northern Mexico, more than 80 percent of the native rangeland is privately owned and is managed primarily as grazingland.

Drought, fire and grazing are the major ecological forces that have historically maintained this area of the southern Great Plains (Sauer 1950, Stebbins 1981, Anderson 1982). Many grassland systems are vulnerable to invasions by woody plants, and climate alone cannot sustain them as grasslands (Sauer 1950). Historically, wildfires retarded or reversed invasions by trees or shrubs, and the grazing of bison and other hooved ungulates also maintained the disturbance pattern that created the habitats upon which grassland birds ultimately depended (Askins 2000, Brennan and Kuvlesky 2005).

While fire once may have been the most important ecological factor in maintaining the grasslands, (Gibson and Hulbert 1987), the introduction of domestic livestock and the development of the ranching enterprise have increased the importance of grazing in determining the nature of these grasslands (Saab et al. 1995).

The pressures of high stocking rates in confined pastures can degrade rangeland habitat over time (Brennan and Kuvlesky 2005). Overgrazing can create conditions that not only preclude fire but facilitate the dispersal and establishment of woody invaders (Risser et al. 1981, Bock and Bock 1987, Humphrey 1987, Steinauer and Bragg 1987, Bock and Bock 1988, Archer 1989). Habitat degradation can be particularly severe where native grazers were historically scarce or absent (e.g., Mack and Thompson 1982, Milchunas et al. 1988, Schlesinger et al. 1990). However, when the grazing of domestic livestock is well-managed, it can be used to improve rangeland habitat for wildlife by simulating historic grazing pressures and patterns (Severson and Urness 1994).

5.4.2 Effect of grazing on grassland birds

Saab et al. (1995) made a comprehensive review of the literature on the effects of grazing on grassland birds. Of 68 species of neotropical migrants in western habitats, 46 percent decreased in abundance with cattle grazing, 29 percent increased, and 25 percent showed no clear response (Saab et al. 1995). A study of 61 riparian bird species in Arizona showed that 53 percent decreased in abundance with cattle grazing, 8 percent increased, and 39 percent showed no clear response (Krueper et al. 2003). In another review, 63 percent of riparian bird species were found to be less abundant in grazed locations, while none were less abundant in ungrazed locations (Tewksbury et al. 2002). While more species respond negatively to grazing than respond positively, there is some variability in the responses, so an analysis of the effect of grazing on neotropical migratory birds must be both habitat-specific and species-specific, and based on field data (Saab et al. 1995).

Rather than responding directly to the presence of livestock, grassland birds usually respond to the effect of grazing and browsing on vegetation (Bock and Webb 1984). Livestock trample plants, may remove some plants entirely, and can cause soil compaction, all of which can change vegetation structure, plant species composition, and vegetation density (Branson 1985, Holechek et al. 1989, Vavra et al. 1994). It is to these structural and floristic alterations that some breeding birds are known to positively or negatively respond (Saab et al. 1995). Birds are affected by habitat structure, floristics and vegetation volume (Willson 1974, James and Wamer 1982, Cody 1985, Mills et al. 1991). Vegetation structure, in particular, is critical to habitat selection and to the productivity of bird species in the grasslands of the central United States (Johnson and Schwartz 1993, McCoy et al. 2001).

Scott et al. (2003) showed that long-term grazing was correlated with a less complex habitat structure and, as a result, with less diverse and abundant populations of 17 bird species in a riparian area in Montana. Long-term overgrazing by livestock can alter vegetation succession by simplifying vegetation structure and composition (Knopf and Cannon 1982, Kauffman and Krueger 1984, Taylor 1986, Schultz and Leininger 1990). This is important because the structural complexity and volume of habitat can influence avian abundance and composition (MacArthur et al. 1962, Willson 1974, James and Wamer 1982, Cody 1985, Mills et al. 1991). Other studies also have demonstrated the negative correlation of grazing pressure with avian abundance and diversity (Kirsch et al. 1978, Taylor 1986, Bock et al. 1993, Ammon and Stacey 1997, Dobkin et al. 1998, Belanger and Picard 1999).

The reduced vegetation density that results from overgrazing has been correlated with increased predation rates on breeding birds (Wray and Whitmore 1979, Johnson and Temple 1990, Clark and Nudds 1991, Riley et al. 1992). Vegetation density is an important predictor of nest predation rates in birds (Bowman and Harris 1980, Martin and Roper 1988), and grazing itself has been associated with higher nest predation rates (Ammon and Stacey 1997).

Grazing is also thought to promote parasitism by brown-headed cowbirds, because the presence of livestock influences cowbird distribution (Lowther 1993, Ortega 1998, Goguen and Mathews 1999, Mayfield 1965; also, see Section 5.5 on Brood Parasitism). Cowbirds are associated with livestock primarily because grazing creates foraging areas for cowbirds (Friedmann 1929, Mayfield, Morris and Thompson 1998). A study of plumbeous vireos (*V. plumbeus*) in New Mexico found that cowbird abundance, brood parasitism of vireo nests (*n*=182), and nests lost to parasitism declined significantly with distance from livestock grazing, regardless of host density or habitat type (Goguen and Mathews 2000).

Overgrazing by cattle can lead to woody plant encroachment into grasslands because it reduces the amount of fuel available for fires (Brown and Archer 1989, Engle et al. 1995) and because cattle effectively disperse the seeds of invasive plants such as juniper (Brown and Archer 1987, Brown and Carter 1998; see Section 5.3 on Vegetation Change).

Grassland birds are decreasing in number more than any other North American avian group (Askins 1993), which is thought to be due to the loss and degradation of grassland habitats

(Knopf 1994, Herkert 1995, Peterjohn and Sauer 1999, Vickery et al. 1999), as well as to fragmentation of habitat patches (Johnson and Igl 2001, Herkert et al. 2003; see Section 5.2 on Habitat Conversion and Land Use Change). While the relative importance of grazing, versus other factors, in the decline of grassland birds is not entirely clear (Clark and Nudds 1991, Knopf 1994), overgrazing practices have been implicated as a primary threat (Saab et al. 1995), as has the invasion of woody plants that often results from overgrazing (Brennan and Kuvlesky 2005).

5.4.3 Overgrazing as a threat to black-capped vireos

There is little quantitative information on the relationships between BCVI populations and a) grazing, b) grazing intensity, and c) types of grazers. But there are comments in the literature suggesting that long-term, intensive grazing is associated with reduced BCVI populations.

Direct effects of grazing and browsing. Black-capped vireos typically construct nests 0.5 to 2.0 m off the ground (range 0.2 to 3.0 m, median 1.0 m) in shrubs or brush mottes where woody vegetation grows close to ground level (Grzybowski 1986, Campbell 1995, Grzybowski 1995). Overbrowsing by goats and white-tailed deer (*Odocoileus virginianus*) removes the low vegetation BCVIs need for nesting (Marshall et al. 1985, Rust and Tazik 1990, Grzybowski 1995), and there is evidence that species that prefer nest sites within 2.5 m of the ground are less abundant when habitat is overbrowsed (Tewksbury et al. 2002).

Early reports suggested that overbrowsing by goats, free-ranging exotic ungulates and white-tailed deer had reduced the habitat available for BCVIs (Graber 1961, Webster 1962). Marshall et al. (1985) stated that one of the major threats to the BCVI was “grazing by sheep, goats, and other exotic herbivores over vast areas of the Edwards Plateau and westward.” Grzybowski (1995) stated that “many areas of the Edwards Plateau are seriously overgrazed by goats and sheep, which remove much potential habitat for use by black-capped vireos.” In the Black-capped Vireo Population and Habitat Viability Assessment Report (USFWS 1996), it was also stated that overbrowsing by sheep, goats and native and exotic wildlife has led to loss of BCVI habitat.

In the Edwards Plateau, goats were specifically implicated in the loss of BCVI breeding habitats. At the time of listing, the Edwards Plateau was the “Angora goat capital of the world,” and according to Marshall et al (1985), there was clear evidence of heavy grazing, trampling and browsing across that region. In a report from Camp Bullis in Bexar and Comal Counties, Shaw et al. (1989) suggested that cattle grazing had little impact on BCVIs, aside from attracting cowbirds, but that grazing by goats was detrimental because they browsed the low hardwood foliage BCVIs use for nesting.

Research at Kerr WMA shows that like white-tailed deer, exotic ungulates prefer green forbs and browse when available (Traweek and Welch 1992). Unlike white-tailed deer, however, many exotic ungulates can shift their diets to grasses when forbs and browse are unavailable, which allows them to compete with domestic goats and, to a lesser extent, with sheep and cattle (Traweek and Welch 1992). As a result, exotic ungulates are likely to have an effect on BCVI habitat similar to that of goats in areas where exotics are increasing and goat numbers are on the decline. When confined within high fences, many exotic species, including axis deer (*Cervus axis*), sika deer (*Cervus nippon)* and blackbuck (*Antilopa cervicapra),* can out-compete white-tailed deer (Traweek and Welch 1992).

While overgrazing and overbrowsing are clearly implicated as a major cause of the loss of BCVI habitat, there is evidence to suggest that the effects are reversible once the overuse ends. For example, at Dobbs Mountain Ranch in Edwards County (in the southwestern Edwards Plateau), Maresh (2004) reported that a history of heavy use by livestock, especially goats, had seriously harmed BCVI habitat. As compared to areas left ungrazed, the overgrazed habitat differed in both structure and composition (Maresh 2003, Maresh 2005a). In areas not subjected to overuse, stands of low-growing shrubs developed substantial foliage in the 0 to 2 m height range, and all known BCVI nests were in these areas before grazing pressure was relieved in the other areas (Maresh 2004). After livestock were removed, the overgrazed sites recovered, there was more vegetation in the 0 to 2 m height range as the browse line diminished, and ground cover increased as grass, forbs and woody species began to re-establish themselves (Maresh 2005a). As a result, the number of breeding BCVI territories in these formerly grazed areas increased from one to seven over a 4-year period (Maresh 2004). Likewise, Sparkman (1996) described habitat that was once grazed heavily by cattle, goats and sheep; once livestock were removed and the area was under less intense browsing pressure by deer, the regrowth of shrubs made it suitable as BCVI breeding habitat.

Indirect effects of grazing and browsing. Overgrazing can cause changes in the vegetation community that indirectly influence the suitability of habitats for use by BCVIs.

Overgrazing may reduce grass cover to the point that prescribed burning is no longer possible and woody cover can develop to the point where habitat is no longer suitable for BCVI (USFWS 1996). In some situations, overgrazing may allow grasslands historically dominated by native warm-season grasses such as little bluestem (*Schizachyrium scoparium*) and indiangrass (*Sorghastrum nutans*) to be overtaken by cool-season grasses such as Texas winter grass (*Stipa leucotrichia*) and prairie dropseed (*Sporobolus asper*). This reduces the fine fuel needed to carry a fire (Tazik et al. 1990).

Another indirect effect is that the presence of cattle may encourage use by brown-headed cowbirds. Research at Fort Hood in Coryell and Bell Counties, Texas suggests a relationship between cattle grazing intensity and cowbird parasitism on BCVI nests (Koloszar and Horne 2000, Kostecke et al. 2003). The number of cattle grazing in one area (9,622 ha) of the base was reduced by 86 percent (from 752 animal units in 1995-1996 to 103 animal units in 1997-1998) while cowbird trapping was being curtailed, to monitor changes in cowbird-BCVI dynamics. After the reduction in stocking rates, cowbirds shifted their foraging to sites where more cattle were present, so that the reported cowbird parasitism rates were 13 times lower than before the number of cattle was reduced. But despite the lower parasitism rates, reducing the cattle stocking rate shifted only the female cowbirds' feeding areas, not their breeding areas. Cowbirds are known to regularly commute up to 7 to 15 km daily (Rothstein et al. 1984, Thompson 1994, Gates and Evans 1998, Curson et al. 2000). Cowbirds appear to be largely regulated by the density and locations of feeding areas (Chace et al. 2005).

The Fort Hood study further suggested that manipulating cattle grazing would be effective only if carried out on a large scale and in the absence of alternative foraging sites for cowbirds (Kostecke et al. 2003). Even with the removal of cattle from certain areas of Fort Hood, there are still ample foraging sites for cowbirds outside the base, considering the rapid urban sprawl in the Texas hill country (Kostecke et al. 2005, Ortega et al. 2005). Nevertheless, there is evidence to suggest that forcing cowbirds to commute longer distances between breeding and foraging sites may reduce parasitism by lowering the fecundity of individual cowbirds (Chace et al. 2005). Cowbirds that commute long distances produce fewer eggs than cowbirds that commute shorter distances and are, therefore, less threatening to BCVIs (Curson and Mathews 2003).

A study at Camp Bullis in Bexar and Comal Counties in central Texas found little parasitism on BCVI nests and suggested that this was because the cattle stocking rate was low, the cattle were free-ranging, and there was no supplemental feeding program that would lure large numbers of cowbirds (Rust and Tazik 1990). However, cowbird parasitism on BCVIs can be relatively high even when there are few or no cattle. For example, at Fort Sill, Oklahoma, where there are no cattle, cowbird parasitism on BCVI nests (before cowbird trapping) was 40 to 50 percent. At nearby Wichita Mountains WR, where cattle and buffalo are present in relatively low numbers, BCVIs were parasitized by cowbirds at rates of 60 to 70 percent (Tazik 1991a).

Some grazing may be compatible with the development and conservation of BCVI habitat as it helps maintain the habitat in an early successional stage. For example, when cattle were removed from some areas of Hill Country State Natural Area in Bandera and Medina Counties, Texas, Sparkman (1996) observed that the resulting successional advancement appeared to contribute to some habitats becoming unsuitable for BCVIs. Grazing by cattle and browsing by white-tailed deer are typically less destructive to BCVI habitat than is grazing by goats and other non-native animals (Guilfoyle 2002).

5.4.4 Trends in livestock numbers

Inasmuch as the effect of overgrazing and overbrowsing by domestic livestock was considered to be a major threat to the species at the time of listing, it is necessary to assess the overall change in livestock numbers since that time to determine whether this threat has increased or decreased. We compared goat and cattle numbers from the USDA Census of Agriculture in 1987 (the year of listing) to the same numbers in 2002, the most recent year for which county-level data are available (2002).

Trends in goat numbers[10] Over the entire Texas and Oklahoma range of the BCVI, there was a 19.1 percent decrease in goat numbers between 1987 and 2002 (Table 5.2). There is a dramatic difference, however, between the two states. Texas had a 22.6 percent decrease in goat numbers, while Oklahoma had a 277.2 percent increase in goat numbers during the same time

---

[10] Unless otherwise noted, the livestock inventories cited here are from the county-level statistics of the USDA Census of Agriculture for the respective years analyzed. These agricultural statistics are available in 5-year intervals. Tables 5.2 and 5.3 present data from 1982 through 2002, but for analytical purposes, we compared only data between the years 1987 and 2002 which were consistent with an overall trend.

span. Nevertheless, goat densities in Oklahoma remain relatively small (one goat per 120 acres) compared to those in Texas (one goat per 35 acres) in the most recent survey (2002). Goat density in Oklahoma was 8.3 goats per 1,000 acres of rangeland compared to 28.7 goats per 1,000 acres in Texas (Table 5.2).

Other regional differences in goat densities can be seen in Figure 5.10 and are described numerically in Table 5.2. In 1987, the Edwards Plateau region had the highest densities of goats (74.3 per 1,000 acres), followed by the North-central Texas region (47.5), the Southwest and Trans-Pecos region (38.7), the Concho Valley (20.9), and the counties of Oklahoma (2.2). In 2002, the Edwards Plateau still had the highest goat densities (48.7), followed by North-central Texas (41.2), the Concho Valley (27.3), the Southwest and Trans-Pecos region (16.1), and Oklahoma (8.3).

The changing patterns of goat densities across the range of the BCVI can be seen in two additional figures. Figure 5.11 depicts the numerical change in goat densities at the county and recovery region levels. Between 1987 and 2002, two regions had net increases in absolute goat densities: the Concho Valley (6.4 goats per 1,000 acres) and the counties of Oklahoma (6.1 goats per 1,000 acres). The remaining regions all saw net decreases in goat densities during this period. The largest decline was in the Edwards Plateau (a decrease of 25.6 goats per 1,000 acres), followed by the Southwest and Trans-Pecos region (-22.6), and North-central Texas (-6.3).

Figure 5.12 depicts the percent change in goat densities at the county and recovery region levels. The counties of Oklahoma had a very large increase in goat densities between 1987 and 2002 (277 percent), while the Concho Valley also had an increase (30.6 percent). The largest decline in goat densities during this time was in the Southwest and Trans-Pecos region (58.4 percent decrease), followed by the Edwards Plateau (34.5 percent decrease), and North-central Texas (13.3 percent decrease).

Most Texas counties had decreasing goat densities between 1987 and 2002, although many counties along the eastern and northern portions of the BCVI's range had stable or increasing goat densities. The largest declines in goat densities have generally been in counties that once had the highest goat densities. All of the counties that had the highest densities in 1987 (Hamilton, Mills, Edwards, Mason, Menard, Kimble, Sutton, Gillespie, Kendall and Uvalde) had moderate to significant declines in goat densities. In fact, seven of the eight counties with the largest numerical decreases in goat numbers between 1987 and 2002 were also counties with the highest densities in 1987.

Trends in cattle numbers. Over the Texas and Oklahoma range of the BCVI, cattle numbers decreased 2.8 percent from 1987 to 2002 (Table 5.3). Despite this slight decrease in cattle numbers overall, significant regional differences can be seen in Figure 5.13 and are described numerically in Table 5.3.

At the state level, Texas had a 9.6 percent decrease in cattle numbers, while Oklahoma had a 12.5 percent increase in cattle numbers. In 1987, the counties of Oklahoma had the highest densities of cattle (323.8 per 1,000 acres), followed by North-central Texas (221.6), the Concho Valley (117.5), the Edwards Plateau (80.0), and the Southwest and Trans-Pecos region (11.9). In 2002, the counties of Oklahoma still had the highest densities of cattle (364.3), followed by North-central Texas (225.7), the Edwards Plateau (74.7), the Concho Valley (63.9), and the Southwest and Trans-Pecos region (8.9).

In both 1987 and 2002, most of the counties with high densities of cattle were located in the eastern and northern portions of the BCVI's Texas range, as well as in most of its Oklahoma range (which, coincidentally, are the areas with the highest cowbird densities and reported parasitism rates). The highest cattle densities (more than 400 cattle per 1,000 acres of rangeland) in Oklahoma were in Blaine, Caddo and Canadian Counties in 1987. These are also the most important counties in Oklahoma for BCVIs. In 2002, cattle densities remained high in these counties as well as in Major, Tulsa and Cleveland Counties. Most of the central and western portion of the BCVI's Texas range had low cattle densities in both 1987 and 2002.

The changing patterns of cattle densities across the range of the BCVI can be seen in two additional figures. Figure 5.14 depicts the numerical change in cattle densities at the county and recovery region levels. Between 1987 and 2002, two regions had net increases in absolute cattle densities–the counties of Oklahoma (40.5 cattle per 1,000 acres) and North-central Texas (4.1). The remaining regions all saw net decreases in cattle densities during this time interval. The most significant decline was in the Concho Valley (a decrease of 53.6 cattle per 1,000 acres), followed by the Edwards Plateau (-5.3), and the Southwest and Trans-Pecos region (-3.0).

Figure 5.14 depicts the percent change in cattle densities at the county and recovery region levels. The counties of Oklahoma had the highest increase in cattle densities between 1987 and 2002 (12.5 percent), while North-central Texas also had a slight increase (1.9 percent).

The largest decline in cattle densities during this time was in the Concho Valley (45.6 percent decrease), while there were also declines in the Southwest and Trans-Pecos region (-25.2 percent) and the Edwards Plateau (-6.6 percent).

Most of the Oklahoma counties and the northern Texas counties had increasing cattle numbers, while most of the central and western portions of the BCVI's range had slightly to moderately (5 to 25 percent) decreasing numbers. In general, the largest decreases in cattle densities occurred in the western portions of the BCVI's Texas range, while the largest increases occurred in the northeastern portion of the BCVI's Texas range, along the southern edge of the range, and in Oklahoma.

5.4.5 Trends in deer numbers

In addition to sheep and goats, "other exotic herbivores" were listed as a major threat to BCVI habitat at the time of listing because these animals remove vegetative cover near ground level that is necessary for BCVI nesting (Ratzlaff 1987). These other herbivores include exotics such as axis deer, blackbuck deer, nilgai antelope (*Boselaphus tragocamelus*), aoudad sheep (*Ammotragus lervia*), fallow deer (*C. dama*) and sika deer. White-tailed deer also remove low vegetation when they browse (Marshall et al. 1985, Rust and Tazik 1990, Grzybowski 1995). As much as is possible, it is beneficial to assess the overall change in white-tailed deer and exotic herbivore numbers since listing.

The number of white-tailed deer in Texas peaked in 1986 at about 4.2 million, but population management and subsequent state regulations resulted in population declines in many areas (Lockwood 2005). Populations in the Edwards Plateau region, however, have remained high and the 2004 population estimate of 1,979,194 deer was the highest on record for the region (Lockwood 2005). In fact, while the Edwards Plateau ecoregion comprised only 28.6 percent of the deer habitat in the state, it harbored 50.5 percent of the state's deer population in 2004. This density of deer results in an average of one deer per 12.07 acres (82.88 deer per 1,000 acres) in the Edwards Plateau of Texas (Lockwood 2005), which exceeds the region's estimated carrying capacity (Harmel and Litton 1981). Statewide, average white-tailed deer density is one deer per 21.3 acres (46.9 deer per 1,000 acres) (Lockwood 2005). The heaviest densities of white-tailed deer in the Edwards Plateau occur in the eastern and central portions of the ecoregion (e.g., Burnet, Travis, Blanco, Hays, Kendall, Comal, Mason, Llano, Gillespie, Edwards, Sutton and Schleicher Counties) (Lockwood 2005).

The Edwards Plateau ecoregion has also traditionally harbored the largest populations of exotic herbivores in the state (Traweek and Welch 1992, Traweek 1995). In the most recent survey of exotics in Texas, approximately 58 percent were found in the Edwards Plateau (Traweek 1995). In the same survey, 62 percent of all confined exotics and 50 percent of all wild exotics were located in the Edwards Plateau (with the highest totals found in Kerr, Real, Edwards, Bandera, Sutton and Medina Counties) (Traweek 1995).

Of the six most abundant species of exotics, all but nilgai showed increasing populations compared to previous surveys (surveys conducted in 1963, 1974, 1979, 1984, 1988 and 1994). In the most recent survey of exotics in Texas (1994), there were 195,483 exotic animals in the state, 77,218 of which were free-ranging (Traweek 1995). Axis deer accounted for 24.7 percent of all confined exotics and 29.3 percent of all free-ranging animals, and had increased in number by 32.9 percent since the 1988 survey (Traweek and Welch 1992). Blackbuck accounted for 21.8 percent of all confined exotics in 1994 and 5.9 percent of all free-ranging exotics, and had increased 43.1 percent in numbers since 1988. Nilgai represented 36.9 percent of the free-ranging exotics (the highest percentage), but very few were confined (0.2 percent of total). Most free-ranging nilgai (97 percent) occur on large ranches in south Texas (Kenedy and Willacy Counties, both outside the range of BCVI), and their numbers have declined 22 percent since the 1988 survey. Between 1988 and 1994, Aoudad populations increased 16 percent, fallow deer increased 42 percent, and Sika deer increased 4 percent (Traweek 1995).

Table 5.2. Number of goats per 1,000 acres of rangeland by county and recovery region within the breeding range of the black-capped vireo.

County	1982	1987	1992	1997	2002	Total change*	% Change**
Oklahoma Counties							
Beaver	0.2	0.3	0.1	0.0	0.3	0.0	13.4
Major	0.1	0.1	2.6	0.2	0.5	0.4	410.1
Tulsa	2.0	1.7	5.3	4.0	19.1	17.5	1058.3
Payne	0.6	2.3	11.6	3.2	11.8	9.5	413.5
Creek	1.2	2.7	2.2	6.2	14.5	11.8	438.7
Blaine	0.0	0.1	0.2	0.2	0.7	0.6	415.3
Dewey	0.0	2.1	0.5	0.4	0.8	-1.3	-62.1
Canadian	0.3	0.8	0.6	1.4	4.4	3.6	454.2
Oklahoma	1.2	0.0	2.5	12.9	27.1	27.1	
Caddo	0.3	8.2	0.0	0.2	0.6	-7.6	-92.5
Cleveland	1.8	13.5	8.2	7.0	28.2	14.7	108.6
Comanche	0.4	0.8	2.0	1.6	2.0	1.2	141.2
Murray	0.4	0.2	0.1	0.4	18.5	18.3	9432.4
Stephens	0.3	1.8	7.0	3.5	6.3	4.4	241.1
Logan	0.8	6.0	6.7	7.8	13.5	7.5	125.2
Kiowa	0.0	2.0	0.0	1.3	0.6	-1.4	-71.2
Garvin	0.3	0.0	1.4	4.0	5.1	5.1	0.0
Grady	0.4	0.9	17.5	1.5	4.0	3.1	332.4
McClain	0.4	2.7	2.1	1.6	3.4	0.7	26.4
Carter	0.5	2.8	0.9	2.3	4.6	1.7	61.5
Jefferson	0.0	0.0	0.3	3.4	7.5	7.5	0.0
Marshall	0.1	1.2	0.8	3.4	6.6	5.4	465.5
Love	0.0	0.9	0.4	2.2	10.3	9.4	885.4
Oklahoma Total	0.5	2.2	3.2	3.0	8.3	6.1	**277.2%**
Recovery region 1: North-central Texas							
Palo Pinto	14.4	7.5	5.7	9.4	9.0	1.5	20.4
Parker	2.1	5.9	20.4	18.8	25.4	19.5	328.0
Coryell	50.2	75.0	73.9	45.6	60.6	-14.4	-19.2
Dallas	10.2	12.9	9.3	28.4	35.7	22.9	178.1
Erath	9.2	19.9	22.1	18.3	32.2	12.3	62.0
Somervell	7.2	33.2	37.5	8.1	25.8	-7.4	-22.3
Comanche	38.3	31.7	43.2	30.0	21.5	-10.1	-32.0
Burnet	49.8	48.1	39.0	39.5	39.5	-8.6	-17.9
Lampasas	37.6	67.9	64.6	65.7	55.6	-12.3	-18.1
Bell	13.7	29.8	26.5	45.0	54.9	25.1	84.3
Brown	54.1	31.5	36.2	30.0	44.7	13.2	41.9
Mills	216.9	324.5	248.0	166.3	170.9	-153.6	-47.3
Williamson	63.0	57.1	43.6	52.3	61.5	4.3	7.6
Travis	12.4	11.0	16.7	10.1	27.8	16.8	152.5
Johnson	2.8	5.0	8.4	14.8	27.3	22.3	441.5

County	1982	1987	1992	1997	2002	Total change*	% Change**
Hood	2.7	4.4	7.6	22.3	31.6	27.2	613.7
Stephens	2.5	5.4	13.9	11.3	8.3	2.9	53.0
Bosque	22.3	49.3	33.1	8.6	20.1	-29.2	-59.2
Hamilton	72.0	121.3	102.6	62.6	49.9	-71.4	-58.9
Coleman	4.1	8.8	9.1	18.8	21.6	12.9	146.5
Region 1 Total	34.3	47.5	43.1	35.3	41.2	-6.3	**-13.3%**
Recovery region 2: Edwards Plateau							
Bandera	38.9	39.7	39.5	42.1	41.0	1.3	3.3
Bexar	4.1	3.4	8.8	16.0	40.1	36.7	1080.6
Medina	1.9	5.7	8.1	9.0	9.3	3.6	63.5
Uvalde	110.0	144.7	173.9	86.1	38.7	-106.0	-73.3
Kinney	52.5	77.1	99.5	71.6	46.2	-30.9	-40.1
Schleicher	41.4	58.0	50.6	43.6	53.6	-4.4	-7.6
Sutton	79.8	100.3	147.9	84.3	69.2	-31.1	-31.0
Blanco	35.7	33.4	76.7	36.3	41.0	7.6	22.8
Hays	24.4	26.6	23.2	45.3	51.4	24.8	93.0
Edwards	134.4	160.9	172.1	105.3	90.7	-70.2	-43.6
Comal	41.4	25.7	40.0	56.2	63.1	37.5	146.0
Gillespie	103.6	124.0	94.2	71.3	73.9	-50.1	-40.4
Kendall	93.6	104.5	95.0	73.9	71.9	-32.7	-31.2
Real	72.2	75.4	89.7	66.9	30.6	-44.8	-59.4
Kerr	42.0	44.1	82.5	41.9	42.5	-1.6	-3.6
Mason	83.3	168.4	90.9	31.4	41.7	-126.7	-75.2
Llano	13.2	19.3	5.6	8.7	10.4	-8.9	-46.3
Kimble	104.7	109.2	97.9	69.2	69.3	-39.9	-36.5
McCulloch	33.3	52.4	42.2	32.6	58.0	5.5	10.6
San Saba	34.5	77.2	30.2	10.9	20.6	-56.6	-73.3
Menard	63.9	110.4	156.2	61.8	60.2	-50.2	-45.5
Region 2 Total	57.6	74.3	77.4	50.7	48.7	-25.6	**-34.5%**
Recovery region 3: Concho Valley							
Coke	10.7	19.3	33.0	24.9	26.9	7.6	39.4
Runnels	4.5	13.1	16.8	26.5	32.5	19.3	147.2
Concho	39.3	54.5	64.2	38.8	66.8	12.3	22.6
Irion	4.1	22.3	20.6	19.9	18.2	-4.1	-18.6
Nolan	7.3	15.8	10.2	2.7	7.2	-8.6	-54.4
Sterling	1.6	6.5	7.8	7.2	5.4	-1.2	-17.7
Tom Green	40.1	31.7	40.5	40.9	52.8	21.1	66.4
Taylor	7.7	4.0	32.6	7.5	8.6	4.6	114.3
Region 3 Total	14.4	20.9	28.2	21.1	27.3	6.4	**30.6%**
Recovery region 4: Southwest and Trans-Pecos							
Brewster	6.0	0.0	0.5	0.4	1.0	1.0	
Terrell	73.8	53.2	42.9	29.7	15.7	-37.5	-70.5

County	1982	1987	1992	1997	2002	Total change*	% Change**
Val Verde	80.8	82.5	90.4	64.2	32.5	-49.9	-60.6
Crockett	27.1	44.0	48.3	29.7	24.5	-19.4	-44.2
Pecos	5.9	14.0	11.8	9.0	6.8	-7.2	-51.2
Region 4 Total	38.7	38.7	38.8	26.6	16.1	-22.6	**-58.4%**
Other Texas Counties							
McLennan	1.6	3.9	3.9	10.9	18.0	14.1	363.2
Reagan	3.2	3.5	2.6	1.4	5.2	1.7	49.2
Lee	0.7	0.6	3.2	4.5	7.9	7.2	1120.6
Bastrop	2.0	2.0	4.8	10.0	13.5	11.5	573.7
Fayette	0.4	0.8	1.1	5.3	5.7	4.9	579.4
Caldwell	1.4	9.5	21.7	22.6	16.7	7.2	75.9
Cameron	3.7	2.7	13.0	9.0	20.3	17.6	653.4
Midland	1.4	3.5	4.6	3.2	10.0	6.5	187.8
Montague	0.5	3.9	1.3	1.7	8.0	4.1	103.8
Grayson	0.9	4.5	6.8	4.9	9.8	5.3	116.0
Cooke	0.7	0.4	3.7	3.5	11.1	10.7	2930.9
Tarrant	4.1	4.7	7.3	23.2	10.6	5.9	125.3
Ellis	2.5	5.0	3.2	10.2	14.8	9.7	192.6
Eastland	12.8	7.8	24.6	19.4	18.1	10.3	132.7
Callahan	2.5	2.2	5.3	7.6	9.0	6.8	306.0
Jeff Davis	0.0	0.0	0.0	0.0	0.1	0.1	0.0
Stephens	0.3	1.8	7.0	3.5	6.3	4.4	241.1
Logan	0.8	6.0	6.7	7.8	13.5	7.5	125.2
Kiowa	0.0	2.0	0.0	1.3	0.6	-1.4	-71.2
Garvin	0.3	0.0	1.4	4.0	5.1	5.1	0.0
Zavala	0.0	31.9	23.0	4.8	11.7	-20.2	-63.4
Other Total	1.9	4.6	6.9	7.6	10.3	5.7	**123.9%**
Texas Totals	29.4	37.2	38.9	28.2	28.7	-42.4	**-22.6%**
TX & OK Totals	29.9	39.4	42.0	31.2	37.0	-36.4	**-19.1%**

* Total change measures the numerical change (positive or negative) in goat density from 1987 (when the BCVI was listed as endangered) to 2002.

** Percent change measures the percentage change (positive or negative) in goat density from 1987 to 2002. It is calculated by subtracting the goat density in 1987 from the goat density in 2002, dividing that value by the 1987 goat density, then multiplying by 100.

Table 5.3. Number of cattle per 1,000 acres of rangeland by county and recovery region within the breeding range of the black-capped vireo.

County	1982	1987	1992	1997	2002	Total change*	% Change**
Oklahoma Counties							
Beaver	144.3	158.7	156.4	193.1	154.3	-4.4	-2.8
Major	255.9	280.0	344.0	356.0	424.0	144.0	51.4
Tulsa	483.7	390.1	322.4	469.7	597.6	207.5	53.2
Payne	329.9	288.0	328.8	337.4	336.1	48.1	16.7
Creek	253.2	224.4	234.3	287.5	273.6	49.2	21.9
Blaine	350.3	489.7	456.2	442.2	446.2	-43.5	-8.9
Dewey	144.2	167.2	172.8	169.7	174.1	6.9	4.1
Canadian	556.8	562.1	608.4	600.1	586.8	24.7	4.4
Oklahoma	314.0	361.6	286.9	367.2	351.5	-10.1	-2.8
Caddo	468.5	448.0	446.7	499.6	491.6	43.6	9.7
Cleveland	539.2	395.1	444.8	486.2	447.4	52.3	13.2
Comanche	283.9	292.1	275.3	294.4	324.8	32.7	11.2
Murray	253.8	245.5	192.1	256.0	266.9	21.3	8.7
Stephens	318.9	290.1	278.4	328.9	358.7	68.6	23.7
Logan	333.6	393.6	404.0	371.6	455.7	62.2	15.8
Kiowa	257.1	317.8	284.6	356.9	310.7	-7.1	-2.2
Garvin	343.4	314.3	305.1	303.8	302.1	-12.2	-3.9
Grady	407.2	402.8	417.7	441.7	441.9	39.1	0.8
McClain	417.7	374.2	371.1	434.6	397.6	23.4	0.2
Carter	235.7	239.7	248.9	287.6	266.2	26.5	0.7
Jefferson	279.8	274.8	230.4	293.7	352.1	77.3	2.7
Marshall	227.6	179.8	203.6	201.1	265.8	86.0	3.0
Love	369.4	358.2	296.0	291.7	353.1	-5.1	2.6
Oklahoma Total	329.0	323.8	317.8	350.9	364.3	40.5	**12.5%**
Recovery region 1: North-central Texas							
Palo Pinto	115.6	116.7	103.8	130.1	125.3	8.6	7.4
Parker	333.5	271.1	311.8	331.4	283.2	12.1	4.5
Coryell	135.3	119.7	132.3	160.1	255.8	136.1	113.7
Dallas	323.2	322.9	295.1	280.9	277.5	-45.4	-14.0
Erath	322.1	398.8	478.0	575.6	541.8	143.0	35.8
Somervell	256.6	197.9	273.8	197.4	126.3	-71.7	-36.2
Comanche	363.2	304.7	362.2	450.4	410.3	105.6	34.7
Burnet	114.1	106.9	108.8	116.8	102.5	-4.4	-4.1
Lampasas	153.3	110.3	122.3	118.8	119.8	9.5	8.6
Bell	347.2	345.3	350.9	298.7	269.2	-76.0	-22.0
Brown	188.9	143.6	171.4	191.3	169.1	25.4	17.7
Mills	154.9	131.4	124.8	153.5	120.9	-10.6	-8.0
Williamson	408.6	340.2	311.3	331.2	278.2	-62.0	-18.2
Travis	184.2	233.8	188.3	118.2	194.4	-39.5	-16.9
Johnson	416.1	507.0	560.7	600.3	396.9	-110.1	-21.7
Hood	202.3	210.6	226.3	262.1	270.2	59.7	28.3

County	1982	1987	1992	1997	2002	Total change*	% Change**
Stephens	87.2	93.3	88.7	94.8	78.5	-14.8	-15.9
Bosque	181.9	157.3	171.3	163.5	166.1	8.7	5.6
Hamilton	201.4	218.1	212.7	240.0	228.0	9.9	4.6
Coleman	126.3	102.2	113.0	123.2	100.4	-1.8	-1.7
Region 1 Total	230.8	221.6	235.4	246.9	225.7	4.1	**1.9%**
Recovery region 2: Edwards Plateau							
Bandera	73.2	60.5	56.9	48.5	44.0	-16.5	-27.3
Bexar	224.3	202.1	255.8	259.9	219.9	17.8	8.8
Medina	179.1	186.5	194.8	146.8	143.9	-42.6	-22.9
Uvalde	73.3	73.2	92.9	89.8	81.2	8.0	11.0
Kinney	30.1	37.0	34.9	23.2	18.9	-18.1	-48.9
Schleicher	43.0	37.9	39.8	33.2	23.8	-14.1	-37.3
Sutton	29.5	26.1	27.4	24.4	15.3	-10.9	-41.6
Blanco	92.2	78.9	81.8	82.9	69.7	-9.2	-11.7
Hays	180.9	121.6	84.4	136.5	139.6	18.1	14.9
Edwards	22.7	20.4	21.2	16.1	18.5	-1.9	-9.1
Comal	138.9	114.6	104.8	117.0	115.8	1.2	1.1
Gillespie	126.0	107.3	99.3	99.9	97.8	-9.5	-8.9
Kendall	93.4	79.5	76.6	71.6	54.6	-24.9	-31.3
Real	44.3	28.0	26.4	24.4	20.4	-7.6	-27.0
Kerr	52.4	48.4	52.9	46.2	45.0	-3.4	-7.0
Mason	108.8	99.3	90.9	99.2	125.8	26.5	26.7
Llano	101.4	99.2	97.8	90.6	86.6	-12.6	-12.7
Kimble	42.6	29.6	32.7	31.4	29.2	-0.4	-1.5
McCulloch	86.9	78.2	87.5	88.1	82.0	3.8	4.8
San Saba	100.9	102.9	103.6	115.2	108.3	5.4	5.2
Menard	50.9	47.6	48.5	44.7	27.8	-19.8	-41.7
Region 2 Total	90.2	80.0	81.5	80.5	74.7	-5.3	**-6.6%**
Recovery region 3: Concho Valley							
Coke	67.6	50.2	47.3	45.4	25.6	-24.5	-48.9
Runnels	156.6	206.4	154.4	144.0	124.3	-82.1	-39.8
Concho	61.0	58.0	54.6	57.9	43.7	-14.4	-24.7
Irion	27.6	25.4	26.0	27.8	16.0	-9.4	-37.1
Nolan	131.6	99.2	98.7	119.3	43.5	-55.8	-56.2
Sterling	25.3	21.1	24.5	22.8	14.5	-6.6	-31.4
Tom Green	74.3	84.1	80.9	80.8	90.2	6.1	7.3
Taylor	293.8	396.0	274.4	285.0	153.7	-242.3	-61.2
Region 3 Total	104.7	117.5	95.1	97.9	63.9	-53.6	**-45.6%**
Recovery region 4: Southwest and Trans-Pecos							
Brewster	19.3	16.2	18.3	16.7	7.9	-8.3	-51.3
Terrell	10.3	4.3	6.6	9.1	5.2	0.9	20.2
Val Verde	8.3	7.4	7.0	11.6	9.3	1.9	25.6

County	1982	1987	1992	1997	2002	Total change*	% Change**
Crockett	16.5	13.7	15.2	13.4	9.5	-4.2	-30.5
Pecos	17.9	17.7	18.4	15.6	12.4	-5.3	-30.1
Region 4 Total	14.5	11.9	13.1	13.3	8.9	-3.0	**-25.2%**
Other Texas Counties							
McLennan	451.9	510.4	462.0	554.9	486.8	-23.6	-4.6
Reagan	17.9	19.6	19.6	19.3	9.7	-9.9	-50.5
Lee	540.6	565.1	624.2	622.4	507.0	-58.1	-10.3
Bastrop	454.1	425.7	455.3	517.2	462.7	37.0	8.7
Fayette	573.1	608.7	557.6	595.2	502.9	-105.8	-17.4
Caldwell	431.5	409.1	364.7	369.8	328.1	-80.9	-19.8
Cameron	314.6	200.2	221.6	240.0	252.0	51.8	25.9
Midland	35.7	33.5	36.3	31.2	24.9	-8.7	-25.9
Montague	252.5	275.4	246.0	295.5	317.6	42.2	15.3
Grayson	444.3	423.1	456.7	460.2	385.9	-37.2	-8.8
Cooke	328.6	283.6	354.4	408.9	410.3	126.6	44.7
Ellis	385.4	403.8	405.6	385.1	322.6	-81.2	-20.1
Eastland	193.6	219.9	184.5	250.6	257.1	37.2	16.9
Callahan	110.8	145.3	138.7	177.9	137.2	-8.1	-5.5
Tarrant	311.5	273.1	296.1	255.6	225.3	-47.8	-17.5
Jeff Davis	23.0	26.2	24.4	23.5	15.9	-10.3	-39.4
Zavala	95.2	101.6	70.6	83.6	94.8	-6.8	-6.7
Jack	119.3	126.0	129.9	139.5	120.4	-5.6	5.2
Wise	383.0	387.9	363.7	432.8	340.1	-47.8	5.1
Denton	349.6	250.6	341.2	407.2	319.2	68.6	4.3
Collin	594.7	524.4	568.9	649.7	434.7	-89.7	8.8
Other Total	305.3	295.9	301.0	329.5	283.6	-12.3	**-4.2%**
Texas Totals	149.1	145.4	145.2	153.6	131.4	-70.0	**-9.6%**
TX & OK Totals	478.1	469.2	463.0	504.5	495.6	-29.6	**-2.8%**

* Total change measures the numerical change (positive or negative) in cattle density from 1987 (when the BCVI was listed as endangered) to 2002.

** Percent change measures the percentage change (positive or negative) in cattle density from 1987 to 2002. It is calculated by subtracting the cattle density in 1987 from the cattle density in 2002, dividing that value by the 1987 cattle density, then multiplying by 100.

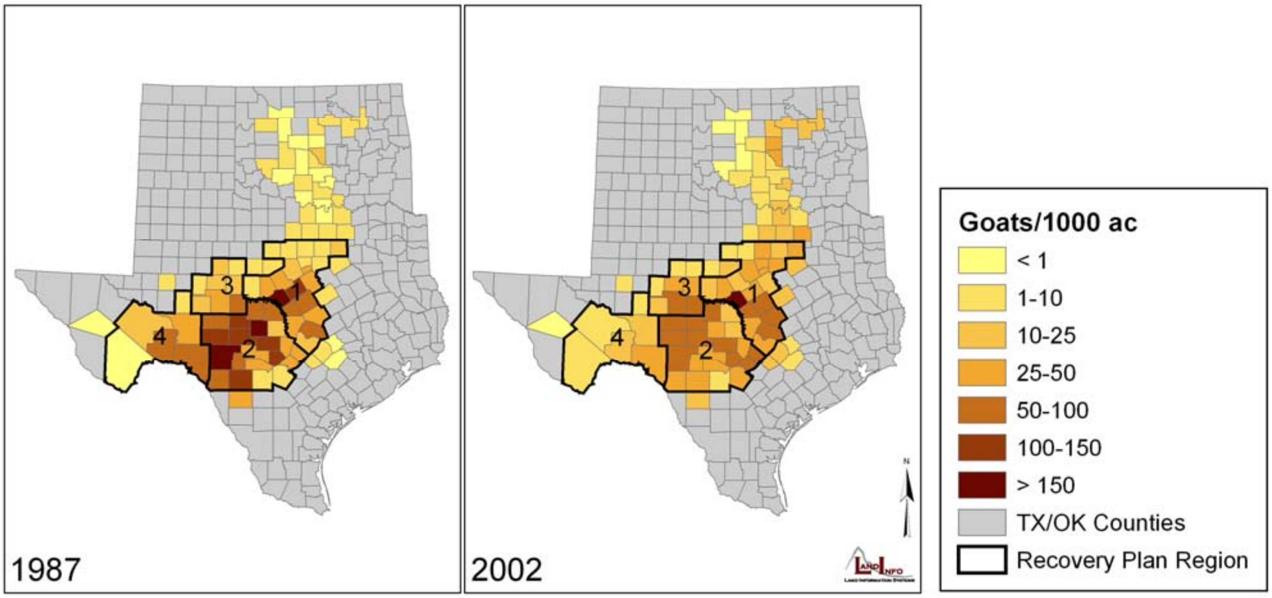

Figure 5.10. Goat densities within the range of the black-capped vireo in Texas and Oklahoma. Goat densities are depicted for both states in 1987 and 2002, as represented by the left and right maps, respectively.

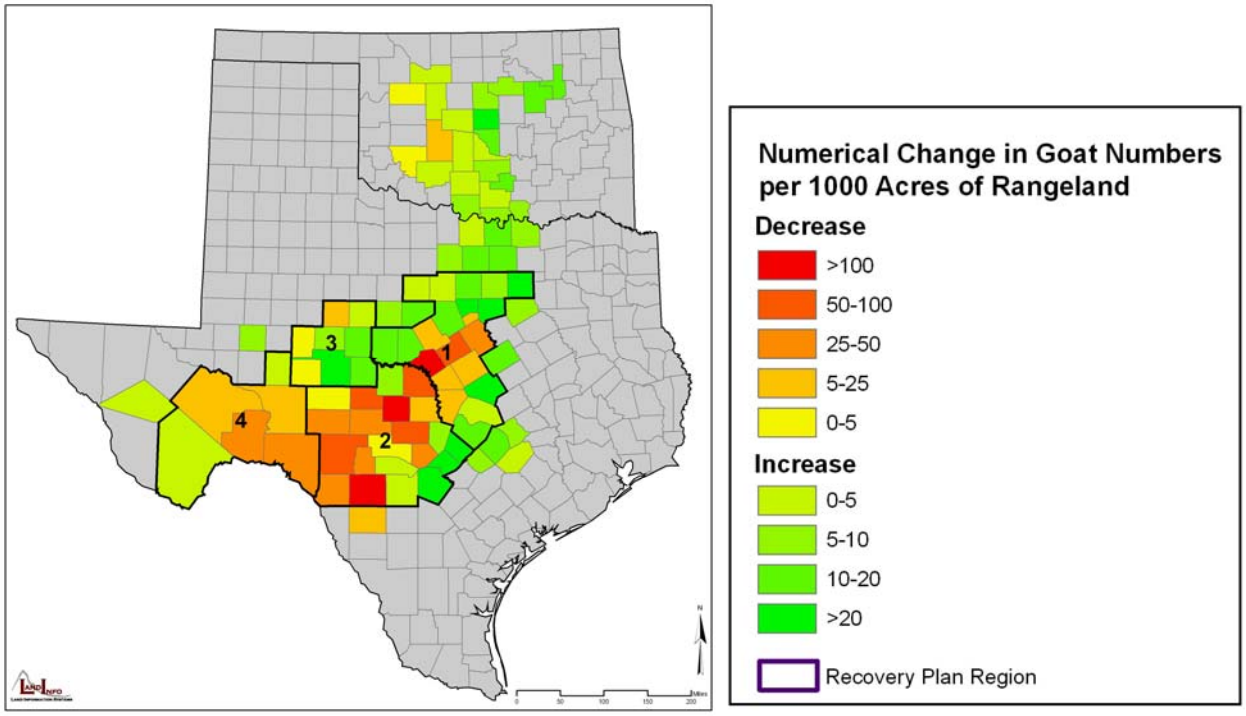

Figure 5.11. Numerical change in goat densities from 1987 to 2002 across the range of the black-capped vireo.

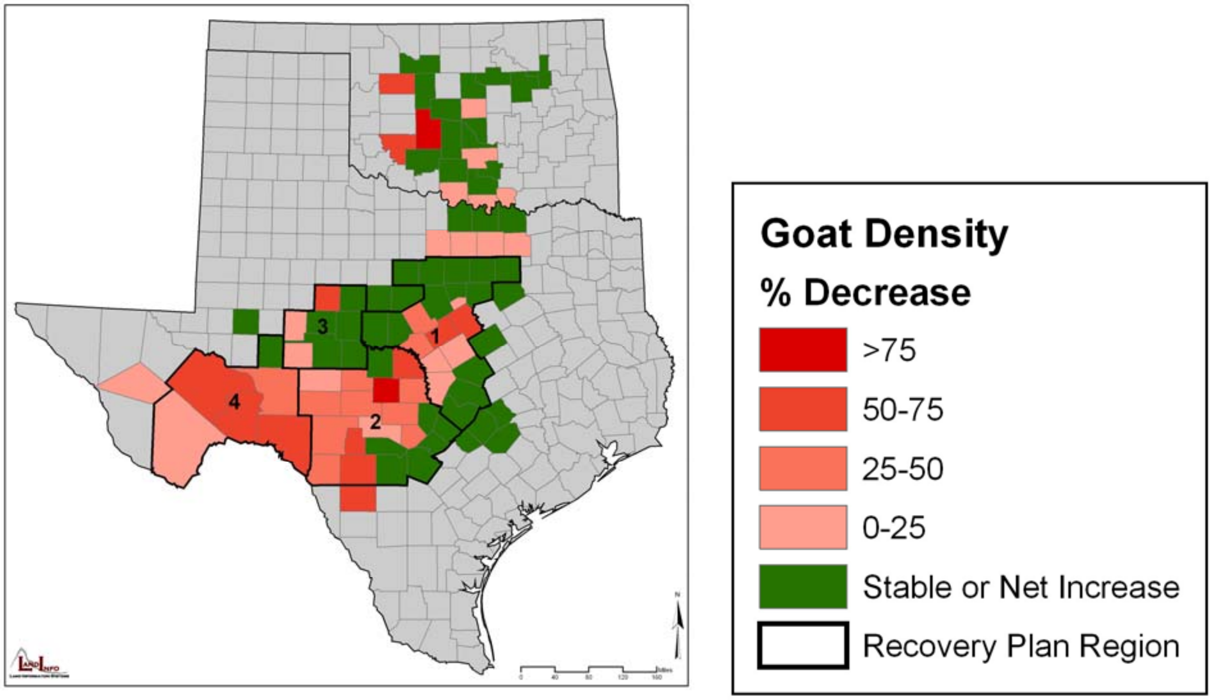

Figure 5.12. Percent change in goat densities from 1987 to 2002 across the range of the black-capped vireo.

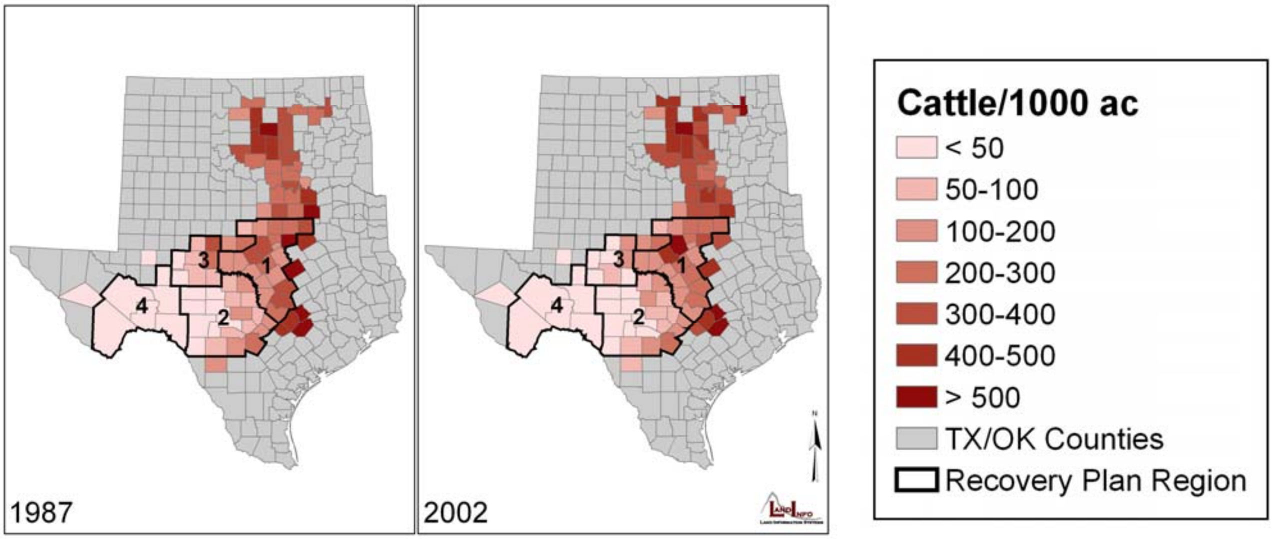

Figure 5.13. Cattle densities within the range of the black-capped vireo in Texas and Oklahoma. Cattle densities are depicted for both states in 1987 and 2002, as represented by the left and right maps, respectively.

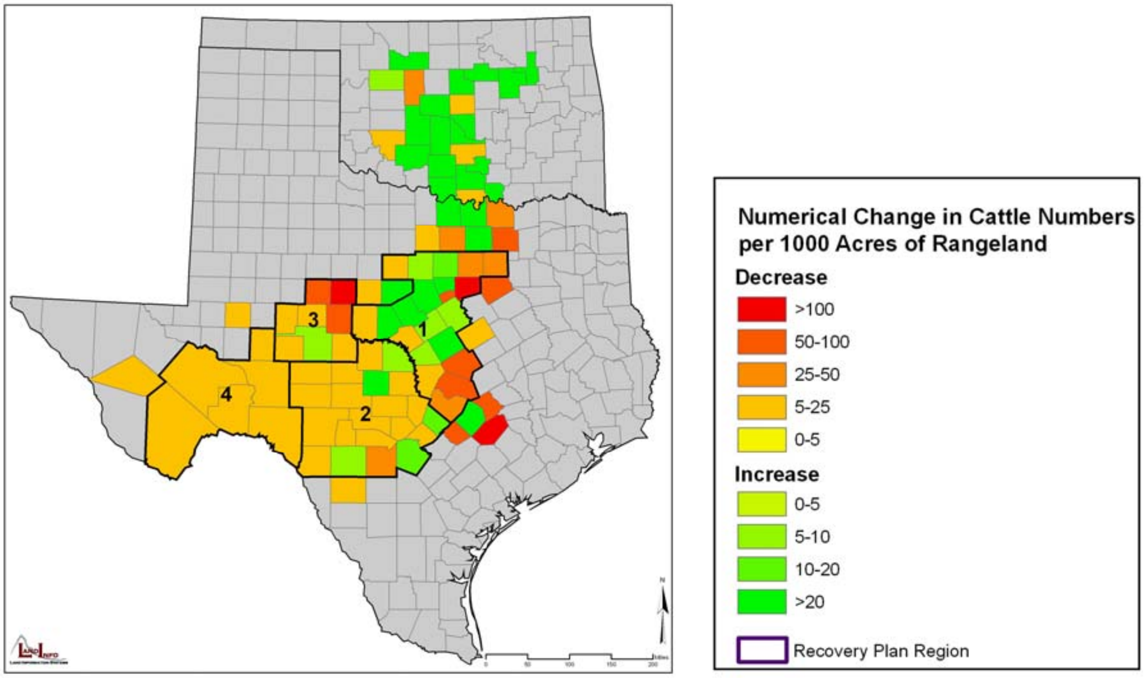

Figure 5.14. Numerical change in cattle densities from 1987 to 2002 across the range of the black-capped vireo.

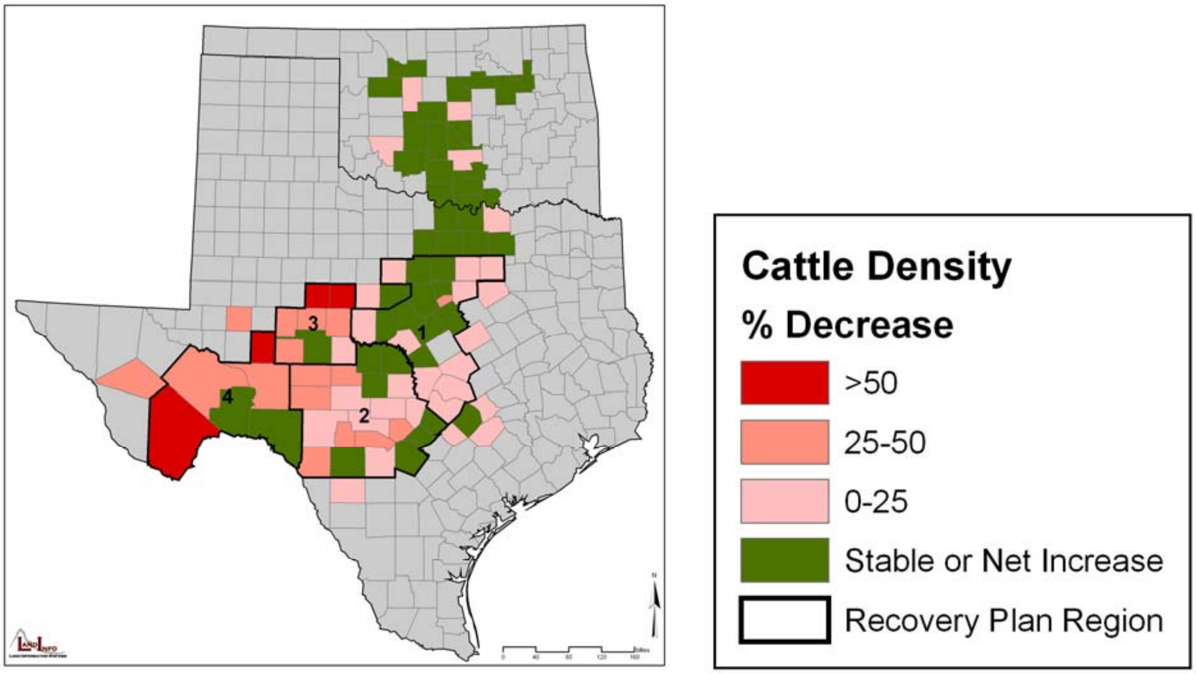

Figure 5.15. Percent change in cattle densities from 1987 to 2002 across the range of the black-capped vireo.

5.5 Brood Parasitism

It is generally accepted that brood parasitism by brown-headed cowbirds has contributed to the declines of several species of songbirds (Brittingham and Temple 1983, Robbins et al. 1989, Terborgh 1989, Robinson et al. 1995). At the time of its listing, the remaining known populations of BCVIs were thought to be seriously threatened by brown-headed cowbirds (Ratzlaff 1987). Brood parasitism rates were reported to be about 80 percent across several study sites in Texas and Oklahoma (Grzybowski 1986).

Here, we assess the changes since listing in the overall threat of cowbird parasitism on the range-wide population of BCVIs. We provide an overview of the most recent research on the ecology of brood parasitism, brown-headed cowbirds, and their effects on BCVIs. Because the biggest factor in the incidence of brood parasitism is cowbird abundance (McGeen 1972, Mayfield 1977, Brittingham and Temple 1983), we include an analysis of cowbird population trends within the range of BCVIs. Our goal is to determine whether the threats to the BCVI from cowbird parasitism have increased, decreased or remained unchanged since listing, and to describe the degree of change, if any.

5.5.1 Ecology of brood parasitism

Brood parasitism is a form of breeding biology, known to occur in birds and insects, whereby the parasites lay their eggs in the nests of host species. The host species cares for the eggs and rears the parasite’s offspring (Davies et al. 1989). Some birds are facultative brood parasites, meaning that they sometimes care for their own young but also lay their eggs in the nests of other birds. Some birds are obligate brood parasites, in which case they never care for their own young. Worldwide, there are approximately 100 species of birds that are obligate brood parasites (Rothstein and Robinson 1998). Three of these species occur in North America– the brown-headed, bronzed (*M. aeneus*) and shiny cowbirds (*M. bonariensis*).

The brown-headed cowbird is the only cowbird that is widespread in the United States, breeding in all states except Hawaii (Lowther 1993). It also breeds in northern and central Mexico. Before the European colonization of North America, the brown-headed cowbird was found primarily in the Great Plains and Great Basin, where it associated with migratory buffalo (Rothstein 1994). Cowbird numbers began to increase dramatically in the eastern United States in the mid- to late 1700s (Mayfield 1965) and in the western states around 1900 (reviewed in

Rothstein 1994). This dramatic expansion east and west of its ancestral range occurred largely as a result of the widespread clearing of forests in the East, Sierra Nevada, Cascades and Pacific Northwest, and because of irrigation and agriculture in the Southwest (Rothstein 1994). These factors improved or created feeding and breeding habitat for cowbirds. It has also been suggested that the historic growth in cowbird numbers may be related to increased winter food supplies (Brittingham and Temple 1983, Robinson et al. 1993). Cowbirds generally prefer habitats that have been altered by humans (Lowther 1993) and are more likely to occur in areas with domestic livestock and where habitat fragmentation has increased the amount of edge (Ortega 1998). Cowbird abundance is known to decrease significantly with increasing forest cover and increase with increasing edge density (meters per hectare) (Donovan et al. 2000, Thompson et al. 2000). Furthermore, nest parasitism levels decrease with increasing forest cover (Hochachka et al. 1999, Thompson et al. 2000) and increase with increasing proportions of open land (grassland and agriculture) (Hejl and Young 1999, Stribley and Haufler 1999).

Some evidence suggests a significant correlation between cowbird and host abundance (Lowther and Johnston 1977, Robinson and Wilcove 1994, Thompson et al. 2000), while other research has not established such a correlation (Robinson et al. 2000). At Fort Hood, Texas, cowbird parasitism of BCVIs was greater where there were denser populations of these potential hosts (Barber 1993, Barber and Martin 1997). This relationship was driven by the density of northern cardinals (*Cardinalis cardinalis*), which were the most abundant bird, as cowbird parasitism of BCVIs was greatest where cardinal density was highest. The density of other co-existing species did not influence this relationship. This correlation suggests that conspicuous hosts, such as the cardinal, might attract cowbirds, which then increases the probability that cowbirds will parasitize any hosts with conspicuous nests or conspicuous behavior, both of which are traits of the BCVI (Barber and Martin 1997).

Cowbirds are unusual among passerines in that they use distinct types of habitat for foraging and breeding and establish separate home ranges for those activities (Chace et al. 2005). Cowbirds forage on the ground in open habitats, often using a variety of feeding sites in modified landscapes such as grazed grasslands, agricultural fields, lawns and livestock corrals (Friedmann 1929, Mayfield 1965, Ortega 1998). There is a particularly strong association between cowbird foraging habitat and the presence of livestock. In a radiotelemetry study in the Midwest, where many alternative foraging sites were available, cowbirds foraged with cattle in

57 percent of the observations (Thompson 1994). Likewise, 98 percent of the foraging observations of radio-tagged cowbirds in a New Mexico short-grass prairie landscape occurred with either pastured or corralled livestock (Goguen and Mathews 2001). Cowbirds commute daily between separate breeding and foraging sites, maintaining regular home ranges in each throughout the breeding season (Rothstein et al. 1984, Thompson 1994, Gates and Evans 1998, Goguen and Mathews 2001).

Several empirical studies have demonstrated that local cowbird abundance declines when the distance from feeding habitat to breeding habitat increases (Morse and Robinson 1999, Tewksbury et al. 1999, Young and Hutto 1999, Goguen and Mathews 2000, Chace et al. 2003). Multivariate models indicate that the distance from feeding sites appears to be one of the most important determinants of cowbird abundance (Tewksbury et al. 1999, Young and Hutto 1999, Goguen and Mathews 2000). However, cowbird abundance does not depend only on the distance to the nearest feeding site, but also on the number of feeding sites within commuting distance, especially in areas where cowbird abundance is low (Chace et al. 2005). For example, local cowbird abundance in Vermont was positively correlated to the number of livestock areas within 7 km, but not influenced by distance to the nearest livestock area (Coker and Capen 1995).

The historic range of the bronzed cowbird extended from the lower Rio Grande Valley of Texas south through Mexico and all of Central America (Sennett 1878, Ridgway 1902, Lowther 1995). Beginning in the early 20th Century, however, its range expanded (Kostecke et al. 2004), and the bronzed cowbird is now a resident in California, Arizona, New Mexico, Texas and Louisiana. Breeding Bird Survey data indicate increasing bronzed cowbird numbers in many of these areas (Sauer et al. 1996, Sauer et al. 2003). Populations in Texas and New Mexico continue to expand northward (Kostecke et al. 2004). In Texas, the current distribution of the species spans most of the southwestern portions of the state and continues northeast to the northern edges of the Edwards Plateau (Bell, Coryell, Eastland and Lampasas counties). Bronzed cowbirds are larger than brown-headed cowbirds and usually parasitize moderate to large passerines (Friedmann and Kiff 1985). They are not generally known to parasitize BCVI nests, although there is one such account on record (Kelly Bryan *in* USFWS 1991).

The shiny cowbird is a South American species that has colonized the West Indies in the past 100 years (Cruz et al. 1985) and has recently invaded Florida (Cruz et al. 1998). Shiny cowbirds have been reported in several other southeastern states from Texas and Oklahoma to the Carolinas, but in small numbers (Grzybowski and Fazio 1991, Post et al. 1993). Because of the restricted ranges of the bronzed and shiny cowbirds, and the fact that there have been few reports of parasitism on BCVI nests, we will focus only on brown-headed cowbirds from this point forward.

5.5.2 Relationship between cowbird abundance and parasitism rates

The threat to BCVIs that we address in this chapter is not cowbirds themselves, but rather, parasitism of BCVI nests by cowbirds. Unfortunately, there are no data on trends in parasitism of BCVIs, but there are data on trends in cowbird abundance (see Section 5.5.3). While cowbird abundance does relate to the threat of parasitism to host species such as the BCVI, the relationship is not linear. And there might be site-specific thresholds, above which parasitism rates do not vary with cowbird abundance.

It is often accepted that cowbird parasitism is directly related to cowbird density at the local scale (McGeen 1972, Mayfield 1977, Brittingham and Temple 1983), but the correlation is not always straightforward (Chace et al. 2005). For example, at forest sites in Illinois, Robinson et al. (2000) found that the percentage of host nests parasitized was not related to cowbird abundance, but the number of cowbird eggs per parasitized nest increased significantly with increasing cowbird abundance. Robinson et al. (2000) suggested that the cowbird-to-host ratio is a more useful index to levels of parasitism than is absolute cowbird abundance because it controls for host abundance. Nevertheless, they acknowledged that the cowbird-to-host ratio is not a good predictor of parasitism levels for all species, especially for the hosts preferred by cowbirds, which is likely the case for the BCVI. In a study of cowbird parasitism across several midwestern landscapes, Thompson et al. (2000) found that cowbird abundance and parasitism levels were positively correlated across study areas at the landscape scale; however, the relationship was not always linear, as habitat fragmentation and other landscape variables can influence parasitism rates, and both cowbird numbers and parasitism levels are often regulated at several spatial scales.

Despite the subtle ways in which this relationship can be influenced by other factors at the local scale, there does seem to be a clear relationship between regional cowbird abundance and the threat of parasitism. Therefore, an overall change in cowbird abundance over time suggests a corresponding change in the threat of parasitism to host species across the same geographic area. It remains unknown whether there is a threshold below which a decrease in cowbird abundance might benefit the host population.

5.5.3 Trends in brown-headed cowbird abundance

Breeding Bird Surveys have revealed an overall downward trend in brown-headed cowbird numbers in Texas since the surveys began (Fig. 5.16), and a less dramatic but also declining trend in Oklahoma (Fig. 5.17). Combining the data for both states, the overall declining trend can be seen (Fig. 5.18). The trend is consistent with overall patterns of cowbird abundance across North America: cowbird populations have been declining since the beginning of the BBS surveys in 1966 (Peterjohn et al. 2000, Wiedenfeld 2000, Sauer et al. 2003). In fact, BBS data indicate that cowbirds declined 1.2 percent per year in North America when averaged across the years 1966 to 2003 ($P<0.01$) (Sauer et al. 2004). During this period, there was a 39 percent decrease in the relative abundance of cowbirds on BBS routes. It should be noted, too, that most of these declines, including those in Texas and Oklahoma, started well before any cowbird control programs began (Rothstein and Peer 2005).

As analyzed by Sauer et al. (2005), the observations of brown-headed cowbirds at the state level trend significantly downward in both Texas and Oklahoma (Figs. 5.16 and 5.17). Likewise, the trend is downward when considering those routes that met our selection criteria for the BCVI's U.S. breeding range (Fig. 5.18, see section 5.3.3. for selection criteria and analytical methods). However, considering pair-wise comparisons at the state level within the BCVI's breeding range, there were clearly some differences between the states. In the Oklahoma range, the mean observations per route of 23.7 (SE=3.6) for the last 10 years (1995-2004) was essentially unchanged from the mean of 23.8 (SE=4.1) observed during the 10-year period before listing (1976-1987). In Texas, however, the mean observations of 15.1 (SE=2.2) over the last 10 years was less than the mean of 20.0 (SE=2.6) observations per route from the 10 years prior to listing ($P<0.05$). This translates to a 23.4 percent decline in the relative abundance of brown-headed cowbirds since listing.[11] During these same periods, the pattern of change in brown-headed cowbird abundance seems to roughly coincide with regional boundaries (Fig. 5.20). For

---

[11] The sample size available for these analyses was large enough to draw conclusions concerning trends at the state level; however, we were not able to reliably assess trends at the recovery region level.

example, relative abundance increased in the Southwest and Trans-Pecos region while apparently decreasing in the Edwards Plateau and Concho Valley regions. In North-central Texas, abundance tended to increase as part of the southern extension of some areas of increase in central Oklahoma. In summary, it appears that the overall abundance of brown-headed cowbirds has declined in major portions of the species' range in Texas and has remained stable in the species' range in Oklahoma.

Although speculative, one possible explanation for the decline is a general pattern of afforestation (Donovan et al. 2000, Thompson et al. 2000). For example, large decreases in cowbird numbers have occurred in southeastern Canada and the northeastern United States where significant reforestation has occurred in the last 100 years (Askins 1993).

While an examination of trends across the states of Texas and Oklahoma (Figs. 5.16-5.18) is revealing, it is also important to describe trends specific to those portions of the states that encompass the actual range of the BCVI. To accomplish this, we estimated annual counts of cowbirds within the BCVI's Texas range, using the same route selection procedure described above and based on methods of James et al. (1996). Of the 70 BBS routes within the Texas range of the BCVI, 31 met the overall selection criteria and are presented in Figure 5.19.[12] Sample sizes for routes that met the selection criteria in Oklahoma were inadequate so we were unable to present a similar figure for Oklahoma.

While overall cowbird numbers have decreased over time, there are some important changes in spatial variability that have occurred, likely in response to changing land use and other factors. Figure 5.20 displays the percent change in breeding season brown-headed cowbird abundance from the period 1976-1987 to the period 1995-2004. In addition to an overall decline in cowbird numbers in Texas and Oklahoma, there has also been a shift in abundance patterns within the two states since the decade immediately before listing (Fig. 5.20). The area of greatest change was in the central Edwards Plateau (depicted in dark green on Fig. 5.20), where cowbird abundance decreased dramatically.

Cowbird trends vary among the recovery regions. In Recovery region 1 (North-central TX, cowbird numbers have largely remained constant. In Recovery region 2 (Edwards Plateau), the overall pattern is a decline in population, although cowbird numbers have increased in some

---

[12] Individual cowbird counts were square-root transformed before computation of annual means, following James et al. (1996).

portions of this region, mainly along the southern border. Cowbird abundance in Recovery region 3 has generally declined. A large portion of Recovery region 4 (Southwest and Trans-Pecos) has shown slight, and sometimes moderate, increases in cowbird numbers.

In Oklahoma, cowbird numbers are generally increasing across the current and historic range of the BCVI. In the counties with current BCVI populations, cowbird numbers have shown moderate increases. Within much of the rest of the historic range, cowbird numbers have remained relatively stable.

The first Mexican record of brown-headed cowbird parasitism on a BCVI nest was reported recently at Rancho La Escondida in Coahuila state (Farquhar and Gonzalez 2005). No formal surveys of cowbirds were conducted at this site, but cowbirds appeared to be less abundant there than in central Texas (Farquhar and Gonzalez 2005). At sites surveyed for BCVIs in Nuevo Leon and Tamaulipas states, both brown-headed and bronzed cowbirds were rarely encountered (Farquhar and Gonzalez 2005). Livestock (e.g., cattle, sheep and goats) grazing in those states typically occurs at higher elevations than the BCVI breeding habitat, which might explain the relatively low abundance of cowbirds at BCVI breeding sites (Farquhar and Gonzalez 2005). However, research results on the impacts of cowbird parasitism in Mexico are limited.

5.5.4 Cowbird parasitism rates

The literature on cowbird parasitism frequently notes alarmingly high rates of parasitism on BCVI nests. It is worth noting, however, that such high rates are not necessarily representative of parasitism across the range of the BCVI or across time.

One example often cited is the case of brood parasitism at Fort Hood, which exceeded 90 percent before the cowbird trapping program began in 1988 (e.g., Tazik 1988, Tazik 1991b, Tazik and Cornelius 1993, Eckrich et al. 1999, Hayden et al. 2000). A parasitism rate of 90.9 percent (*n*=33) was found in 1987 and a rate of 90.8 percent (*n*=87) was found in 1988, for a combined rate of 90.8 percent (*n*=120) over the 2-year period. Data gathered elsewhere support the notion that natural parasitism rates on BCVI nests are high. Between 1983 and 1985, the overall parasitism rate on BCVI nests at three sites in Texas (*n*=33) and one in Oklahoma (*n*=3) was 86 percent (Grzybowski 1985a). Cowbird parasitism on BCVI nests at Kerr WMA in Texas fluctuated from 82 percent (*n*=33 nests) in 1986 to 66 percent (*n*=6) in 1987 and to 90 percent (*n*=10) in 1988 (Grzybowski 1991). A 1987 study of all known BCVI sites in Oklahoma found a parasitism rate of 85 percent (*n*=14) across four counties (Grzybowski 1987).

However, not all BCVI populations have faced such a high cowbird parasitism rate. For example, a study at Kickapoo Caverns State Park documented a nest parasitism rate of 24.7 percent (20 of 81 nests) in 1992 (Keddy-Hector 1992). At Devil's River State Natural Area, the parasitism rate was 48 percent without cowbird control, but BCVIs were still able to maintain 50 percent nesting success (*n*=93 territories) in 1990 (Bryan and Stuart 1990). Farquhar and Maresh (1996) noted that parasitism is generally lower in the southwest portion of the BCVI range. Between 2001 and 2004, at three sites in the western portion of the Texas range (Big Bend National Park in Brewster County, Chandler Independence Creek Preserve in Terrell County, and Dobbs Mountain Ranch in Edwards County), the overall parasitism rate of monitored BCVI nests (*n*=75) was 28 percent (Maresh 2005a). At Dolan Falls Preserve in Val Verde County, natural parasitism rates on BCVI nests were 32 percent (*n*=25 nests) in 2000 and 27 percent (*n*=30 nests) in 2001 (Farquhar, unpublished data).

In general, it appears that the highest rates of parasitism occur in the northern and eastern portions of the BCVI's breeding range. The highest reported parasitism rates have occurred in Oklahoma (85 percent; Grzybowski 1987) and in the northeastern part of the Texas range (more than 90 percent at Fort Hood; Tazik 1991b), while rates have been much lower at sites farther away from the ancestral cowbird range, such as at Kickapoo Caverns State Park (24.7 percent; Keddy-Hector 1992) or Devil's River State Natural Area (48 percent; Bryan and Stuart 1990), both of which are located in the southwestern Edwards Plateau of Texas. Others have suggested that parasitism rates on BCVI nests might be low in Mexico as well (Grzybowski 1995, Farquhar and Gonzalez 2005).

These results support the notion that cowbird abundance and parasitism frequency on most host species generally decline with distance from the historic (and current) center of cowbird abundance in the northern Great Plains (Hoover and Brittingham 1993, Smith and Myers-Smith 1998, Thompson et al. 2000). For example, wood thrush (*Hylocichla mustelina*) parasitism levels are typically more than 80 percent in Illinois (Hoover and Brittingham 1993, Robinson et al. 2000) but less than 25 percent in Maryland (Dowell et al. 2000, Petit and Petit 2000). Similarly, parasitism of grassland birds is generally higher in the northern Great Plains than in the Midwest (Davis and Sealy 2000, Koford et al. 2000). Therefore, at the continental scale, cowbird abundance is often a good predictor of parasitism levels (Chace et al. 2005).

Cowbird parasitism can be quite variable over time. Parasitism rates for southwestern willow flycatcher (*Empidonax trailii extimus*) nests at various sites in Arizona ranged from 8 percent to 21 percent between 1994 and 1996. Along the South Fork of the Kern River in California, parasitism rates fluctuated from 50 percent to 80 percent from 1989 to 1997; and in the Gila River Valley of New Mexico, they ranged from 14.7 percent to 27 percent between 1995 and 1997 (Whitfield and Sogge 1999). In another study of willow flycatchers across several study areas, parasitism rates varied from 10.9 percent to 40.7 percent over a 10-year period (Sedgwick and Iko 1999). Parasitism rates on least Bell's vireos (*V. bellii pusillus*) at the San Luis Rey River in California varied from a low of 19 percent to a high of 56 percent over a 9-year study period (Kus 1999). A study of the effect of cowbirds on plumbeous vireos in Colorado found parasitism rates that ranged from 38 percent to 66 percent over a 13-year period (Chace and Cruz 1999). In a 7-year study of warbling vireos (*V. gilvus*) in Colorado, parasitism rates fluctuated from 40 percent to 100 percent across the years (Ortega and Ortega 2003).

At Kerr WMA in Kerr County, Texas, the reported parasitism rates on BCVI nests fluctuated from 65 percent in 1985 (*n*=20) to 90 percent in 1988 (*n*=10) (Grzybowski 1991). Across several sites in Oklahoma, parasitism on BCVI nests decreased from 92 percent in 1986 (*n*=13) to 58 percent in 1987 (*n*=19), while for several sites in Texas, parasitism changed from 76 percent in 1986 (*n*=37) to 53 percent in 1987 (*n*=15) (Grzybowski 1988). Such variability in cowbird parasitism from year to year must be considered before making generalizations concerning the region-wide impact of parasitism.

Parasitism rates also can be highly variable across space within a breeding season (Robinson et al. 1995). For example, across sites in Arizona, mean annual parasitism rates for southwestern willow flycatchers ranged from 3 percent to 48 percent, and across sites in California, parasitism rates ranged from 29 percent to 66 percent (summarized in USFWS 2002). Populations within short distances of each other also can experience very different parasitism rates. Three sites at Malheur NWR in Oregon were all located in the same drainage, had similar types of vegetation, and were within 2 km of each other, but had mean parasitism rates that ranged from 15.4 percent (range of 11.1 to 87.1 percent) at one site to 18.8 percent (0.0 to 53.1 percent) at another to 41.5 percent (10.9 to 40.7 percent) at the third, over a 10-year period

(Sedgwick and Iko 1999). This level of variability further demonstrates the need to be cautious about extrapolating the results from one study across space and time.

The ecological costs of cowbird parasitism are not reflected entirely by measures of nest success. In their analysis of studies of cowbird parasitism, Lorenzana and Sealy (1999) noted that 95 percent of empirical studies defined the cost of parasitism in terms of the difference in the number of young fledged from parasitized vs. unparasitized nests (i.e., “percent parasitized”). This approach does not necessarily reflect the selection pressure that parasitized individuals face because sometimes they renest and can raise the same number of young as unparasitized individuals (Smith 1981, Roth et al. 1996). This approach is, therefore, of limited value as compared to actually quantifying the productivity of females over the entire breeding season with a measure such as seasonal fecundity (Pease and Grzybowski 1995, Lorenzana and Sealy 1999, Grzybowski and Pease 2000).

The effect of parasitism rates on BCVIs cannot be considered separately from predation and other factors influencing recruitment. It is important to understand the difference between the effects of parasitism on individual host nests vs. its effect on entire host populations. Although parasitism almost always reduces the reproductive success of parasitized nests, the host species often produce enough young to maintain stable populations (Rothstein 2004). Likewise, in some portions of the species range, predation rates (e.g., snakes and fire ants) appear to increase as cowbird parasitism is decreased. Because there are no available data on post-fledging survival for BCVIs, there remains some uncertainty as to whether predation on post-fledglings could likewise vary under varying parasitism rates.

5.5.5 Effects of cowbird parasitism on host populations

Brown-headed cowbirds are known to parasitize at least 220 bird species, 144 of which are known to raise cowbird young (Friedmann and Kiff 1985, Lowther 1993). Cowbirds are host generalists and can parasitize several co-occurring passerine species (Friedmann and Kiff 1985, Carter 1986, Wiley 1988). Because female cowbirds do not select only a single host species (Friedmann 1963, Fleischer 1985, Hahn et al. 1999), the decline of a particular host species will not produce a corresponding decline in cowbird populations (Rothstein 1975, Mayfield 1977, Grzybowski and Pease 1999). Thus, cowbirds are a relatively greater threat to host species that already have small populations because of other factors. As a result, cowbird parasitism has been deemed a primary threat for several endangered species or subspecies such as Kirtland's warbler (*Dendroica kirtlandii*) (Mayfield 1977), BCVI (Grzybowski et al. 1986, USFWS 1991), least Bell's vireo (Goldwasser et al. 1980, Franzreb 1989a), and southwestern willow flycatcher (Unitt 1987, Brown 1988), as well as for common species such as dickcissel (*Spiza americana*) (Fretwell 1977).

Brood parasitism is typically a proximate threat as opposed to an ultimate threat, with many other factors usually contributing to the overall threat to the species. In fact, the recovery plan for the BCVI recommends an integrated approach to managing land, habitat and grazing as a way of reducing the long-term effects of cowbird parasitism (USFWS 1991). Furthermore, brood parasitism generally affects seasonal fecundity (young raised per female per year) in passerine birds less than nest predation does (Grzybowski and Pease 2000).

When considering the results of 44 separate studies on 25 host species, the effect of cowbird parasitism on the success of individual nesting efforts becomes clear–significantly fewer young are fledged per nest (Lorenzana and Sealy 1999). What is not so obvious, however, is the overall effect parasitism has on the population parameters of the host species (Robinson et al. 1995). For example, in prairie warblers (*D. discolor*), Nolan (1978) found that while parasitized nests resulted in 74 percent fewer young, the overall effect on the population was only a 13 percent reduction in seasonal fecundity. They found that prairie warblers routinely abandon their parasitized nests and then renest. Similarly, Sedgwick and Knopf (1988) found that the effect of cowbird parasitism on the overall seasonal productivity of willow flycatchers was much less than one would expect from simply measuring the productivity of individual nests. While the success of parasitized nests was low (18.2 percent vs. 56.3 percent for unparasitized nests), the overall nest success for parasitized pairs was at least 54.5 percent because renesting is a successful strategy with lower parasitism rates than initial nests. Wiedenfeld (2000) examined large-scale patterns of abundance (BBS data) for cowbirds and eight species of wood warbler host species in the eastern United States and Canada and found no relationship between cowbird abundance and population changes in the host species.

In their response to brood parasitism, host species are typically classified as either egg "rejectors" (sometimes called "ejectors," Rothstein 1976, Scott 1977) or egg "acceptors." Rejectors (Rothstein 1975) remove cowbird eggs from their nests or simply abandon the parasitized nests altogether (e.g., Clark and Robertson 1981, Sealy 1992). Acceptors do not remove cowbird eggs from their nests or abandon them (Rothstein 1975). Grzybowski and Pease (1999) further divide acceptors into two categories: insensitive acceptors (species that raise cowbird young but are not detrimentally affected by this at the population level) and extinction-prone acceptors (species that are detrimentally affected by cowbird parasitism).

The cowbird incubation period is only 11 days, so cowbird nestlings usually hatch before the host's young and cause the death of some or all of the host's young (Rothstein 2004). Particularly hard hit are species with incubation periods longer than 10 to 12 days, such as the small vireos and the small flycatchers (Ehrlich et al. 1988, Briskie and Sealy 1987). The BCVI has an incubation period of 14 to 17 days (Graber 1961). Cowbird hatchlings out-compete the much smaller BCVI hatchlings, which soon die (Graber 1957, Grzybowski et al. 1986). Few parasitized BCVI nests are successful. Parasitized nests at Kerr WMA produced 0.2 fledglings per nest in 1986-1987 (Grzybowski 1995), and parasitized nests at Fort Hood (n=13) fledged 0.9 BCVIs per nest (Tazik 1991b). In fact, at Kerr WMA, where cowbird parasitism was reduced to 3% of observed nests, BCVI seasonal fecundity increased from 0.9-1.0 young/female/season to 3.78 (Grzybowski 1995). Graber (1961) observed no BCVI young fledged from parasitized nests at a site in Oklahoma, and 40 percent of all eggs laid were lost to cowbird activity. However, 60 percent of BCVI pairs were still able to fledge at least one young during the nesting season, be it from a first or second nesting attempt.

Some host species respond to parasitism by deserting their nests (Clark and Robertson 1981, Sealy 1992). Black-capped vireos desert parasitized nests at a much higher rate than unparasitized nests, and respond to nesting failure by renesting, even after successful broods, and by remating during an extended breeding season (Graber 1961, USFWS 1991, Tazik and Cornelius 1993, Hayden et al. 2000). At Fort Hood, 42 percent (42 of 101) of observed nests with one cowbird egg were abandoned in favor of a second nest (Tazik 1991b). In most populations studied in Oklahoma and Texas, only about 10 percent of females fledged two broods, although when cowbird parasitism levels were reduced to 3 percent of observed nests, 25 percent of females successfully fledged two BCVI broods in a season (Grzybowski 1995). At Fort Hood, very few BCVIs had more than two successful nesting attempts in a season (only four of 170 pairs in 2003 and nine of 202 pairs in 2004) (Cimprich 2003, Cimprich 2004).

Female cowbirds often remove a host egg from nests they parasitize (Ortega 1998). This has been observed at Fort Hood (Stake and Cavanagh 2001), where video monitoring also documented cowbirds removing BCVI nestlings from seven of 133 (5.3 percent) BCVI nests (Stake and Cavanagh 2001). In two of those cases the entire brood (four nestlings) was removed, and two nestlings were removed on four occasions. The same study at Fort Hood documented three instances of cowbirds ingesting or removing fecal sacs produced by host nestlings.

Cowbird parasitism has also been shown to skew the sex ratios of host offspring in song sparrow (*Melospiza melodia*) nests (Zanette et al. 2005). The presence of cowbird young increased competition within the nests; female fledglings were at a competitive disadvantage because of their smaller size and had high rates of mortality, reducing the proportion of female sparrows in the parasitized nests to half that of the unparasitized nests in the study. Parasitism can also alter adult sex ratios by increasing the mortality of nesting females, who must expend much more energy to tend parasitized nests (Robinson et al. 1995). In one study, the ratio of adult female BCVIs to adult males was significantly higher after cowbird control reduced parasitism levels to less than 30 percent, compared to the same population where parasitism had exceeded 80 percent before control (Grzybowski unpublished data). Females may also disperse from heavily parasitized areas. Over a 4-year period, it was found that the annual return rates for male BCVIs were constant, while female return rates were twice as high (89 percent vs. 45 percent) following a year with low parasitism rates (4 percent) as they were following a year with high parasitism rates (45 percent) (Grzybowski 1991).

A female cowbird can deposit 30 to 40 eggs (Ankney 1985, Robinson et al. 1995, Rothstein et al. 1986, Scott and Ankney 1983) into the nests of ten or more hosts per year (Grzybowski and Pease 1999). Most of these eggs have no effect on host productivity because they are placed in the nests of egg ejectors or are lost to predation (Rothstein 1977, Robinson et al. 1995). For example, at Dolan Falls Ranch Preserve, approximately 70% of parasitized BCVI nests in 2000-2001 (*n*=37) were subsequently depredated (Farquhar et al. 2005). A female cowbird normally lays two to eight eggs in nests of appropriate hosts (Hahn et al. 1999). According to DNA analyses, the potential mean annual reproductive capacity (i.e., fecundity) of female cowbirds is effectively 2.8 eggs (ranging from one to 13 with SD=2.7) (Alderson et al. 1999). A 6-year study of cowbirds in Manitoba, Canada found similar results, with a realized female fecundity (defined as eggs laid in host nests that could potentially fledge cowbird young) of only 2.3 ± 0.6 (*n*=59, range of one to 13) (Woolfenden et al. 2003).

5.5.6 Cowbird removal

Trapping effort. Since listing, there has been a substantial cowbird trapping effort by state and federal agencies and private landowner groups. Verifiable records of the last 5 years show that an average of 47,589 individuals are removed per year (most of them females) across the BCVI's range.[13]

The areas in Texas with the largest trapping efforts (more than 5,000 birds trapped) include Coryell (the location of Fort Hood), Burnet, Blanco, Bandera, Kerr, Kimble and McLennan Counties (Fig. 5.21). In Oklahoma, the only trapping effort occurs in Comanche County at Fort Sill and Wichita Mountains WR.

Since 2000, 228,660 cowbirds have been trapped in the state of Texas and 9,286 in Oklahoma. The largest effort has been in Coryell and Bell Counties, Texas, where a total of 66,726 cowbirds have been trapped since 2000. Of those, 53,495 were caught at Fort Hood (Summers et al. 2000, Summers and Sterling 2001, Summers and Norman 2002, Summers and Norman 2003, Summers and Norman 2004). Most of these cowbirds are likely migrants and only 25 percent are potential local breeders (Kostecke et al. 2005). It is possible that a large percentage of cowbirds trapped elsewhere are also not local breeders.

Regionally, the southern portions of Recovery region 1 (Travis, Williamson, Burnet, Lampasas, Coryell and Bell Counties) are areas where moderate to high levels of trapping are carried out. Less trapping occurs in other parts of the region.

In Recovery region 2, the most intensive trapping occurs in the center of the region (Kimble, Kerr and Bandera Counties) and in Blanco County. Moderate trapping generally occurs in the southwestern (Edwards, Kinney, Real and Uvalde Counties) and northeastern (San Saba, Mason and Llano Counties) portions and in Kendall County. There is little trapping elsewhere in the region.

There is also little cowbird trapping in Recovery region 3. Only 147 cowbirds have been trapped in Tom Green County during the last 5 years, and fewer than 100 in the other counties.

---

[13] This includes trapping programs operated by Fort Hood, Fort Sill in Oklahoma, Wichita Mountains WR in Oklahoma, and private landowners in Texas who are required to report their numbers to the Texas Parks and Wildlife Department (Linda Campbell, personal communication).

The same is true of Recovery region 4. In Val Verde County, a total of 2,237 cowbirds were trapped between 2000 and 2004. The other counties in the region have each trapped fewer than 100 cowbirds.

In Oklahoma, cowbird trapping has occurred only in Comanche County, where it has been intense. Between 2000 and 2004, 12,104 cowbirds were trapped at Wichita Mountains WR (personal communication, S. Waldstein) and 2,283 cowbirds were trapped at Fort Sill (personal communication, G. Wampler). There are no known cowbird removal programs in Mexico (Farquhar and Gonzalez 2005).

Influence of cowbird removal on cowbird abundance. At the regional and range-wide levels, there does not seem to be any clear association between cowbird trapping and cowbird abundance. In Comanche County, Oklahoma, cowbird trapping has been intense, yet the numbers of cowbirds are increasing moderately. Much of Recovery region 3 in Texas has seen dramatic decreases in cowbird numbers, but there is little trapping there. In Recovery region 4, where there also has been little trapping, cowbird abundance has remained relatively constant across the region. The areas of dramatic declines in cowbird numbers in Recovery region 2 include some counties with little or no trapping, some counties with moderate trapping, and only one county (Kimble) with intensive trapping. The other counties in this region where the trapping effort has been intense (Kerr, Bandera and Blanco) show moderate decreases in cowbird numbers. In Recovery region 1, the areas with intensive cowbird trapping (Burnet and Coryell Counties) have had only slight decreases in cowbird numbers, as have most of the counties with moderate levels of cowbird trapping. The only exceptions to this are in Somervell County and the western portion of Johnson County, where moderate levels of trapping coincide with slight increases in cowbird numbers.

Thus, it is unclear whether cowbird trapping has led to declines in cowbird numbers across the range of the BCVI. In fact, the area of the Edwards Plateau with the sharpest declines in cowbirds is composed almost entirely of counties with only low to moderate trapping efforts. The exception to this is in Kimble County and part of Kerr County, where cowbird trapping is intense and cowbird numbers are on the decline. One might suggest that trapping efforts in Kimble and Kerr Counties have been successful enough to affect the cowbird populations in adjacent and nearby counties. If this were the case, however, we would expect the same to be true of Coryell and Bell Counties, where the largest trapping program in the state (at Fort Hood) is based. This does not appear to be the case, however, because cowbird numbers in these and adjacent counties have largely remained constant. At Fort Hood, the number of cowbirds killed each year has not declined since the control program began (Eckrich et al. 1999, Summers and Norman 2004). While the evidence is overwhelming that cowbird trapping programs can reduce nest parasitism at the local level, there is little evidence that trapping has helped reduce cowbird numbers at a regional or range-wide level. These patterns are consistent with research on Kirtland's warblers (DeCapita 2000) and least Bell's vireos (Griffith and Griffith 2000), in which increased trapping success had little or no effect on cowbird numbers.

Influence of cowbird removal on parasitism rates. It has been suggested that parasitism is a concern when it affects more than 30 percent of a host population (Halterman et al. 1999). This level was set to indicate a significant impact because studies by Mayfield (1977) and Laymon (1987) showed that a 30 percent parasitism rate may make a host population unstable. However, the Mayfield paper also noted that ovenbirds (*Seiurus aurocapillus*) reproduced well despite 50 percent parasitism. Grzybowski and Pease (2000) demonstrated through modeling that the relationship between percent parasitism and seasonal reproductive success (seasonal fecundity) is complex, and that 30 percent of nests parasitized is probably too low to be a threshold of concern for most host species. They suggest that passerines can often tolerate parasitism exceeding 50 percent. Based on this information, and the fact that parasitism rates can be variable in space and time, Smith (1999) suggests that managers should consider implementing cowbird management programs only when parasitism rates in a local sample of 30 or more nests exceeds 50 percent over a time span of at least 2 consecutive years. Likewise, Tazik (1991b) suggested that a parasitism rate of 50 percent on BCVI nests at Fort Hood could serve as the upper limit for an acceptable rate that would allow the BCVI to maintain stable populations and still allow for incidental losses from military activities.

Intensive cowbird trapping has been associated with dramatic reductions in cowbird parasitism for many species. Parasitism rates of Kirtland's warbler nests dropped from 70 percent between 1966 and 1971 to only 6 percent from 1972 to 1977, after cowbird trapping (Bocetti 1994). Least Bell's vireos at Camp Pendleton, California suffered parasitism rates of about 50 percent in the early 1980s before cowbird trapping. The rate dropped to between 4 percent and 20 percent from 1983 to 1987 and to no more than 1 percent since 1988 (Griffith and Griffith 2000). Mean parasitism rates on southwestern willow flycatchers in California dropped from 63 percent (1989-1991) to 17 percent (1994-1997) following intensive cowbird control (Whitfield et al. 1999).

Brood parasitism of BCVI nests at Fort Hood, Texas exceeded 90 percent (e.g., Hayden et al. 2000) before the initiation of cowbird control in 1988. The control program at the base has coincided with reduced cowbird parasitism levels and increased nesting success for BCVIs (Eckrich et al. 1999, Kostecke et al. 2005). Similarly, cowbird trapping programs elsewhere in Texas and Oklahoma have dramatically decreased parasitism rates on BCVI nests at those sites.

The control program at Fort Hood was relatively ineffective until 1991 when biologists began to target pastures with large concentrations of cattle, adopted innovative trap designs, manipulated trap numbers, and began shooting female cowbirds (Eckrich et al. 1999). The switch to targeting pastures with large concentrations of cattle occurred because it was thought that the base contained too much host breeding habitat for trapping to be cost-effective. Between 1988 and 1990, parasitism rates at the base remained above 50 percent, but they fell to just 8.6 percent by 1997. The number of breeding male BCVIs increased from 85 (Tazik and Cornelius 1993) to 357 (Koloszar 1998) over the same 10-year span. Intensive cowbird trapping has continued since that time, and parasitism on BCVI nests has remained low, with overall mean annual rates ranging from 5.4 percent to 7.4 percent between 1999 and 2004 (Summers et al. 2000, Summers and Norman 2002, Summers and Norman 2003, Summers and Norman 2004). Overall, the parasitism rate has not exceeded 10 percent since 1997. The mean number of territorial male BCVIs has increased significantly, and nest success has had a strong negative correlation with parasitism rates across Fort Hood (Kostecke et al. 2005). While researchers at Fort Hood largely attribute BCVI population increases to cowbird control efforts, they acknowledge that this would not have worked in the absence of ample early-successional habitat (Kostecke et al. 2005).

Between 1987 and 2004, researchers at Fort Hood found a strong negative correlation between the number of female cowbirds trapped during the BCVI breeding season and the incidence of cowbird parasitism on BCVI nests (Summers and Norman 2004, Kostecke et al. 2005). An experiment was conducted in 1997, in which cowbird trapping was stopped temporarily. The result was increased cowbird densities and parasitism levels, and decreased reproductive success of BCVIs (Cook et al. 1998). However, it is difficult to assess the true effects of cowbird trapping on BCVI success at Fort Hood because the effects are confounded by other, simultaneous management actions, such as manipulations of cattle grazing (Kostecke et al. 2005).

At several study sites in the Edwards Plateau between 1983 and 1990, cowbirds parasitized 72 percent of BCVI nests where no cowbird removal occurred, but only 24 percent of nests in areas where cowbirds were removed (Grzybowski 1990b). In the same study, BCVIs produced only 0.4 young per female per year in areas without cowbird control, but more than 2.0 young with cowbird control during the same sampling period.

Cowbird trapping across all known BCVI locations in Oklahoma in 1986 reduced parasitism from 85 percent (*n*=14) to 50 percent (*n*=2) (Grzybowski 1987). At Kerr WMA, cowbird parasitism of BCVI nests between 1985 and 1988 was 77 percent (*n*=69 nests) in areas without cowbird control, but only 18 percent between 1985 and 1991 (*n*=160) in areas with cowbird trapping (Grzybowski 1991). Between 1983 and 1987, at several sites in Oklahoma and Texas, parasitism rates on BCVI nests were 74 percent (*n*=35) in Oklahoma and 73 percent (*n*=91) in Texas without cowbird control, but only 44 percent (*n*=34) in Oklahoma and 36 percent (*n*=84) in Texas with cowbird control (Grzybowski 1988).

Influence of cowbird removal on host populations. While cowbird trapping programs decrease nest parasitism rates locally, the remaining question is whether this translates into an effect at the population level. Because cowbird trapping programs have often been implemented in conjunction with other conservation measures–primarily habitat enhancement–it is difficult to draw conclusions about the influence of cowbird removal alone. The BCVI population has undoubtedly increased dramatically at Fort Hood since the initiation of cowbird control efforts (Kostecke et al. 2005). But whether or not this can be attributed directly to cowbird management remains to be seen, because it is difficult to assess the individual effects of trapping, shooting or cattle grazing manipulations on parasitism when these management practices were implemented simultaneously (Kostecke et al. 2005). The positive effect of wildfires at Fort Hood also must be considered. In February 1996, crown fires burned 4,015 ha (9,917 acres), including 508 ha (1,255 acres) of BCVI habitat (Goering 1998, Hayden et al. 1999). The abundance of BCVIs has increased on burned areas, while remaining constant on unburned areas (Cimprich 2002). Therefore, it remains to be seen whether the increases in BCVI populations at Fort Hood (or anywhere else where management involves multiple practices) can be directly attributed to cowbird control.

Case studies with other species tend to suggest that the influence of cowbird removal is at least secondary to habitat issues. In one case, cowbird trapping that began in 1972 in Michigan to help the Kirtland's warbler was successful in reducing the nest parasitism rate from 70 percent in 1966-1971 to 6 percent in 1972-1977. The mean number of warbler fledglings per female increased from 0.80 to 3.11 over the same time period (Bocetti 1994). Yet, the warbler breeding population remained relatively unchanged (around 200 pairs) for the succeeding 18 years, despite continued cowbird control efforts. The Kirtland's warbler population finally experienced a dramatic increase beginning in 1990, following a major fire that enhanced the species' habitat (Kepler et al. 1996). The scientific consensus had been that cowbird parasitism was limiting the population of Kirtland's warblers, but in the end, an increase in available habitat produced dramatic population increases that nearly two decades of cowbird trapping had not (Mayfield 1978, Mayfield 1983).

The case of the Bell's vireo is also illustrative. In the late 1970s, the largest population of Bell's vireo occurred in Santa Barbara County, California (Goldwasser et al. 1980, Greaves 1987). Despite an aggressive cowbird trapping program, this population began to decline in the late 1980s and the number of breeding pairs decreased by 60 percent. This was largely attributed to the successional advancement of the riparian vegetation (Rothstein 2004). Kus and Whitfield (2005) argued that cowbird control for least Bell's vireo populations will increase the species' abundance only as long as suitable habitat is available to support population growth.

Similarly, cowbird trapping programs to help the southwestern willow flycatcher in California, Arizona and New Mexico have done little to benefit those populations (Rothstein 2004, Kus and Whitfield 2005). In most cases, trapping has increased the breeding production of southwestern willow flycatchers but has not increased the number of breeding birds (Rothstein et al. 2003). Even today, after years of trapping for cowbirds, known flycatcher territories in California number only about 200 (Kus et al. 2003). It appears that habitat loss has had a much larger effect on the southwestern willow flycatcher than has cowbird parasitism (Rothstein 2004).

Finally, an assessment of the real threat of cowbird parasitism to BCVIs requires more careful analysis than a simple enumeration of nesting loss from brood parasitism and/or the possible benefit of cowbird removal. The fact that the entire geographic range of the BCVI is within the cowbird's ancestral center of abundance in central North America (Mayfield 1965, Hahn and O'Connor 2002) suggests that the species has some adaptations for coexisting with cowbirds (Hall and Rothstein 1999, Hayden et al. 2000, Rothstein 2004). While cowbird control appears to be a reasonable stop-gap for preventing the extirpation of some populations, there is little evidence, for any host species, that cowbird control can actually increase the size of host populations (Rothstein and Peer 2005).

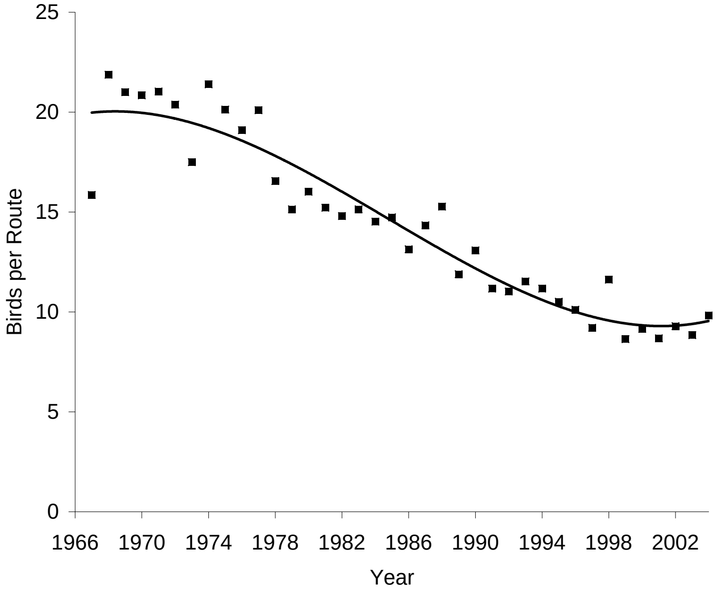

Figure 5.16. Relative abundance trend for brown-headed cowbirds in Texas based on Breeding Bird Survey data (Sauer et al. 2005). There has been a significant downward trend (trend estimate= -2.72, P<0.001, n=193).

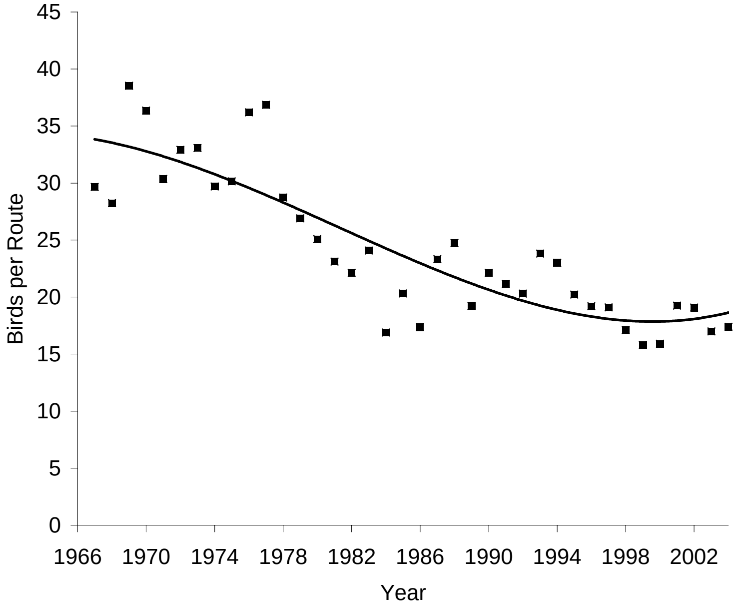

Figure 5.17. Relative abundance trend for brown-headed cowbirds in Oklahoma based on Breeding Bird Survey data (Sauer et al. 2005). There has been a significant downward trend (trend estimate= -1.93, P<0.001, n=60).

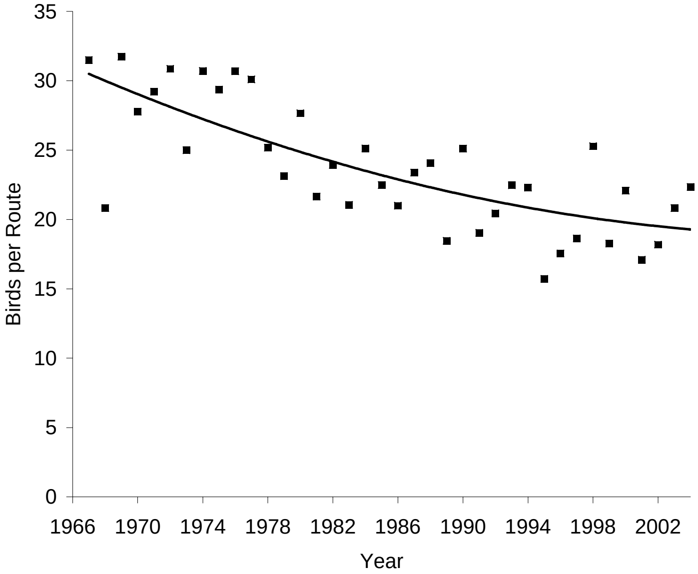

Figure 5.18. Relative abundance trend for brown-headed cowbirds in the U.S. breeding range of the black-capped vireo based on Breeding Bird Survey data (Sauer et al. 2005). Annual counts were estimated using a route selection procedure based on James et al. (1996). Of the 89 BBS routes within the range of black-capped vireos, 40 met the overall selection criteria.

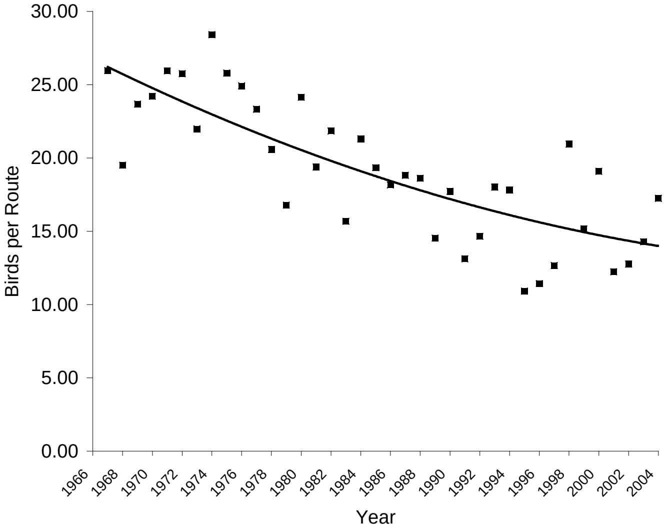

Figure 5.19. Relative abundance trend for brown-headed cowbirds in the Texas section of the breeding range of the black-capped vireo based on Breeding Bird Survey data (Sauer et al. 2005). Annual counts were estimated using a route selection procedure based on James et al. (1996). Of the 70 BBS routes within the Texas range of black-capped vireos, 31 met the overall selection criteria. Individual counts were square-root transformed before computation of annual means. Points presented represent back-transformed data.

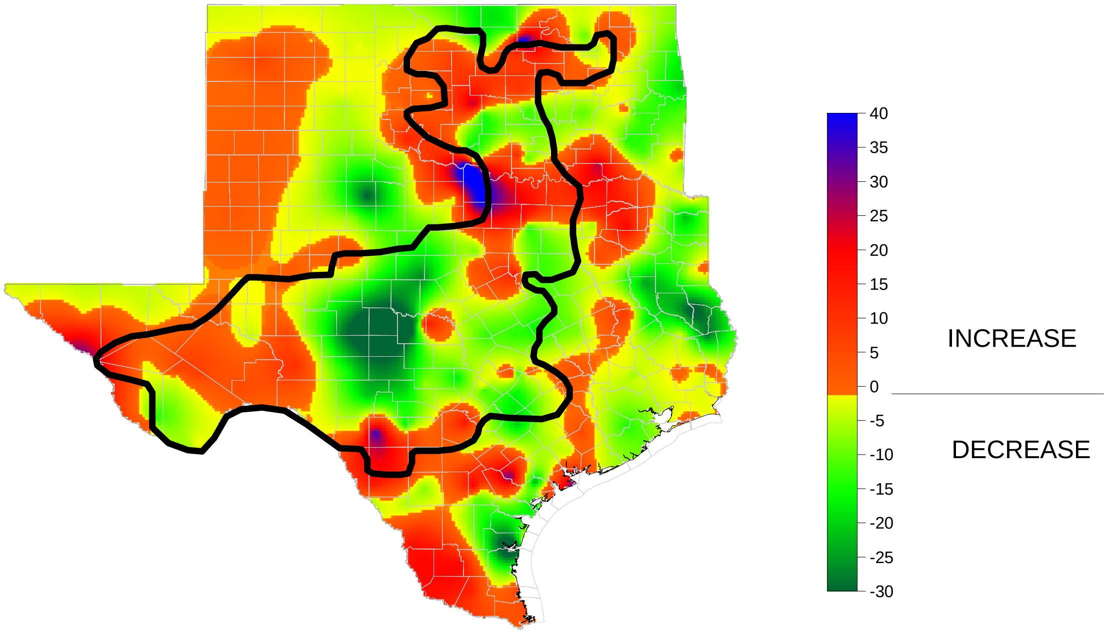

Figure 5.20. Percent change in breeding season brown-headed cowbird abundance from the period 1976-1987 to the period 1995-2004, based on Breeding Bird Survey data (Sauer et al. 2005). The black line represents the estimated U.S. breeding range of black-capped vireos. Mapped values are smoothed data (via kriging) representing differences in relative abundance values between the two periods.

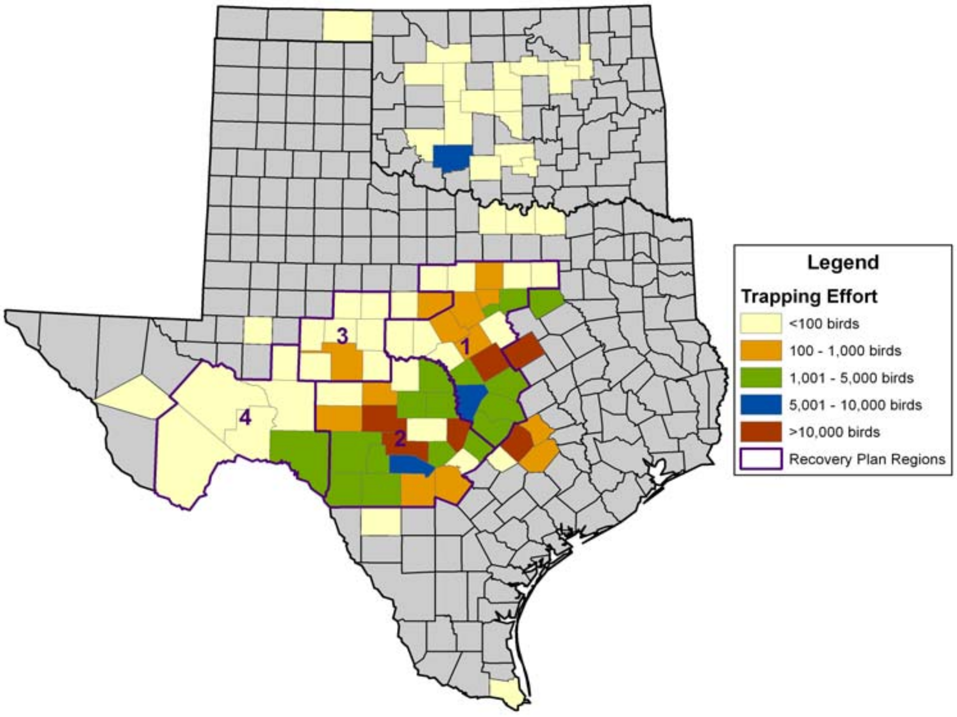

Figure 5.21. Brown-headed cowbird trapping efforts in the range of the black-capped vireo, 2000-2004. Numbered areas reflect current black-capped vireo recovery regions.

6.0 Management and Recovery Efforts

Beyond the active management programs in place at the four major population centers (Fort Hood Military Reservation (Texas), Kerr WMA (Texas), Wichita Mountains WR (Oklahoma), and Fort Sill Military Reservation (Oklahoma)), there are other programs and recovery efforts that may benefit BCVI populations.

Historic response to invasive woody plants has included large-scale brush removal projects that, at times, resulted in the removal of all broad-leaved, low, woody vegetation. While clearing rangelands of all shrubs can be detrimental to BCVIs, selectively controlling juniper, particularly in areas where juniper has recently invaded otherwise suitable breeding habitat, can enhance BCVI habitat while improving rangeland productivity. A recent NRCS Section 7 consultation and Biological Opinion for the implementation of the 2002 Farm Bill suggests a more progressive approach to brush management that protects and enhances BCVI habitats (USFWS 2004).

Under the authority of the recent Section 7 consultation and Biological Opinion, the NRCS can fund brush management projects for up to 50,765 acres of suitable habitat (3.5 percent of the total estimated suitable habitat) throughout the range of the BCVI in Texas (USFWS 2004). This project authority is distributed among the four recovery regions in proportion to the estimated amount of suitable occupied habitat in each. The program helps landowners who develop approved wildlife management plans with juniper removal and other practices such as brush management, prescribed burning, prescribed grazing, and range planting through cost-share agreements. With continued funding, the program could lead to a long-term increase in suitable habitat for the BCVI. As part of the agreement, brush management contractors are required to receive training and certification to operate in and around BCVI habitat. While brush control in BCVI habitat may have some short-term impacts on the species, the long-term benefits should be improved habitat conditions at the landscape level (USFWS 2004). This program has potential for conserving the species, although the participation rates and ultimate results are not yet documented.

Environmental Defense (ED) has initiated a program for establishing Safe Harbor Agreements with private landowners participating in habitat management programs likely to benefit black-capped vireos in 37 counties in Texas. Since December 2000, ED has enrolled 7 landowners accounting for 6,201 acres under the program.

Other BCVI habitat restoration efforts are occurring at several managed areas throughout the Edwards Plateau (USFWS 2004). The Nature Conservancy (TNC) is conducting prescribed fires and removing juniper at the Barton Creek Habitat Preserve in Travis County. The Nature Conservancy is also working to expand BCVI habitat at Dolan Falls Preserve in Val Verde County, Love Creek Preserve in Medina County, and Independence Creek Preserve in Terrell County. Habitat improvement efforts at Cedar Ridge Preserve in Dallas County have led to the first record of BCVI in that county since 1997 (Marsden 2005).

In Travis County, the City of Austin, the Lower Colorado River Authority, and the county itself all have land management programs focused on habitat restoration and management for BCVI at several sites. The Texas State Soil and Water Conservation Board's Brush Control Program is also developing management strategies that target areas where brush control is most needed. Between 2000 and 2004, this program spent approximately $3.6 million per year on brush control (USFWS 2004). Funding for this program was cut to about $600,000 in 2005.

One effort on private lands includes state agencies, federal agencies, non-governmental organizations, and university partners in monitoring and managing the habitat of endangered species throughout the Leon River basin in Coryell and Hamilton counties. Point surveys conducted across this 350,000-acre project area in 2003-2005 yielded baseline estimates that 5 to 7 percent of the area may be occupied by BCVIs (Juarez et al. 2004, N. Wilkins unpublished data). The project is actively enrolling landowners in management contracts to enhance rangeland, including selective juniper removal, grazing deferment and prescribed burning treatments. The project has recently become associated with a Department of Defense effort to provide incentives for the conservation of endangered species, including BCVIs, in an off-site conservation program for nearby Fort Hood.

While there are other locally-led efforts that have implications for BCVI recovery, there is not yet an inventory of these programs along with their associated impacts on BCVIs. This appears to be an informational need that might be addressed through future efforts.

7.0 Summary and Conclusions

The following section is a review of the most relevant findings related to the important conservation issues affecting black-capped vireos. The quality of available data, and therefore the strength of implications, varies with geography and topic. Throughout this analysis, it is apparent that a lack of data for many of the key issues makes it difficult to be direct and definitive in answering the final questions required for this status assessment. We nevertheless offer the following conclusions with the qualifications that are inherent to drawing inference from secondary data sources.

7.1 Geographic Range

At the time of listing, the black-capped vireo was known to have breeding populations distributed across four counties in Oklahoma, 21 counties in Texas, and in Coahuila, Mexico. The historic breeding distribution was thought to include an area stretching from Kansas southward through central Oklahoma and through west-central Texas, with a southern limit in central Coahuila, Mexico. While the overall survey effort has increased since listing, the effort has not been evenly applied across the species' potential breeding range. Even though the resulting data provide incomplete knowledge of the species breeding range, the accumulated results do provide some insight into the conservation status of the species. These results are summarized below.

- Since listing, breeding populations have been documented in 49 Texas counties, five Oklahoma counties and three Mexican states.

- The current black-capped vireo breeding range no longer appears to extend northward past central Oklahoma, and the species has not been documented in Kansas since the 1950s.

- Survey efforts since 2000 have confirmed that there are occupied breeding habitats in 38 counties in Texas, three counties in Oklahoma, and three states in Mexico.

- Recent survey results confirm that the black-capped vireo's breeding range extends substantially farther southward in Mexico than was known at the time of listing. The recent discovery of the southernmost breeding populations of the species significantly expands the known breeding range.

- Given recent observations in the wintering range of the species, black-capped vireos are now known to migrate to wintering habitats along a narrow range stretching from approximately 16 to 27 degrees latitude along the mountainous Pacific coast of Mexico. Recent observations suggest that most of the birds winter in the northern two-thirds of this area.

7.2 Habitat Availability

As identified when the species was listed, the amount and distribution of suitable breeding habitat was a major factor contributing to the species' endangerment. However, at the landscape level, the amounts and distribution of suitable breeding habitat were unknown at the time of listing. Despite some significant progress in refining species-habitat relationships and in estimating the area of potential breeding habitat, there is not yet an inventory that would make it possible to reliably estimate trends in suitable breeding habitat.

- Analysis of extensive roadside surveys conducted in 1996-1998 resulted in an estimate of 1.45 million acres of potential breeding habitat in 53 counties across the species' range in Texas. This amounts to 3.3 percent of the total land area within the counties considered. Due to sampling issues, this estimate lacks reliability and is of limited utility for assessing the species' status.

- The suitability of rangeland as breeding habitat for black-capped vireos largely depends on the composition and structure of woody shrubs and small trees. Habitat alteration by invasive junipers appears to be a major limitation in the maintenance and development of suitable breeding habitats in many portions of the species range.

- The influence of prescribed fire in maintaining habitat suitability in the eastern portion of the species' range appears to be more important than was generally expressed at the time of listing. Although the absence of wildfire "under natural conditions" was acknowledged in the listing decision as a factor in the successional advancement of suitable habitats, the successful application of prescribed fire in managing for black-capped vireos was not well documented or generally acknowledged at the time of listing.

- Fire interacts with a region's physical features and climate to produce different outcomes. Fire appears to contribute to the development of suitable breeding habitats in Oklahoma and the eastern portion of the species' Texas range. However, in the western portion of the species' breeding range in Texas and in Mexico, fire is not as important in maintaining habitat suitability.

7.3 Population Status

At the time of listing, the population status of black-capped vireos was largely established from survey efforts that yielded a known population of 191 pairs (Marshall et al. 1985). Extrapolating from their surveys, Marshall et al. (1985) expanded their survey results to estimate that there were more than 20 pairs in Oklahoma, 188 to 374 pairs in Texas, and 48 to 131 pairs in Mexico. By 1996, about 1,803 males were reported in the U.S. (USFWS 1996); by 2005, the known U.S. population was 5,996 males (This report). Including the breeding range in Mexico, the current known population is at 6,269. Important points concerning population status are summarized below:

- From available survey data it is clear that the overall breeding population of black-capped vireos is substantially larger than was known at the time of listing. However, it is not clear how much of the difference can be attributed to increased survey effort. Because of unequal survey efforts across the species' range, we cannot reliably estimate what proportion of the total breeding population is represented by the current known population.

- In many local cases, it could be that increased survey efforts alone have resulted in larger known populations of black-capped vireos. In other cases, however, it appears that breeding populations have likely increased since listing. For example, known breeding populations in three of the four areas with the most intensive survey efforts have increased almost 10-fold since surveys were reported in 1996 – these include Fort Hood Military Reservation (Texas), Wichita Mountains WR (Oklahoma), and Fort Sill Military Reservation (Oklahoma).

- To date, about 75 percent of the known population in the breeding range is found on four well-surveyed areas– Fort Hood Military Reservation (Texas), Kerr WMA (Texas), Wichita Mountains WR (Oklahoma), and Fort Sill Military Reservation (Oklahoma). Together, these facilities cover approximately 400,000 acres (161,877 ha) – an area representing only 1 percent of the total area of rangeland in the Texas/Oklahoma range of the species. The remaining 25 percent of the known population is the product of documented occurrences from at least 52 other properties, many of which are on private lands with only recent survey access.

- The current known breeding population in Mexico represents only 4 percent of the total known population. However, suitable breeding habitats in Mexico have been only sparsely surveyed, and most of the known breeding range has not been assessed for black-capped vireo occurrence. Where surveys have been conducted in Mexico, black-capped vireos are often found at densities higher than in the species' U.S. breeding range.

7.4. Analysis of Threats

At the time of listing, the identified major threats to the black-capped vireo included habitat loss through land use conversion, vegetation succession, grazing and browsing by domestic and wild herbivores, and brood parasitism by brown-headed cowbirds. While the relative importance of individual threats appears to have changed since listing, these remain the primary threats to the species.

7.4.1 Habitat conversion and land use change

When proposed for listing, the largest known concentration of black-capped vireos was in the immediate vicinity of Austin, Texas and the population was under immediate threat from development and road construction. Much of the subsequent development in the Austin area was mitigated through habitat conservation plans and the subsequent set-aside of mitigation lands, including the Balcones Canyonlands NWR. Habitat conversion and changes in land use continue to pose a threat throughout parts of the species' range. There are no data available for directly measuring trends in the amount of suitable habitat for the species, but some overall changes in land ownership and land use do suggest indirect trends that might be important for black-capped vireo conservation. However, these data were available only for the U.S. portion of the bird's breeding range.

- As of 2002, approximately 80 percent of the 68.8 million acres in the species' U.S. breeding range was classified as farm and ranchland. About 70 percent (33.9 million acres) of farm and ranchland in the area was classified as rangeland. This represents the land base on which suitable habitat for black-capped vireos might presently exist or be developed, either through management or natural processes.

- Recent trends in land use, land ownership and land fragmentation in the Texas part of the breeding range are quite different than those in Oklahoma.

- Over the period 1992-2002, the total area classified as rangeland declined by 8.6 percent across the breeding range in Texas. This apparent change in land use was partly driven by an overall loss in farm and ranchland, but the reported loss of rangeland was 37 percent more than the overall loss in farm and ranchland. The figures collected for Oklahoma suggest a stable or slightly increasing inventory of rangeland. All else being equal, a net loss in rangeland area likely represents a loss of potential habitat for the species – but the magnitude of loss as well as compensating factors are unknown.

- Over the period 1992-2002, about 2.8 million acres of large farms and ranches (more than 2,000 acres) were broken into smaller ownerships across the species' range in Texas. Oklahoma experienced a slight increase in large ownerships during the same period.

- During this same period, the numbers of smaller farms and ranches increased by about 40 percent across the species' breeding range in both states.

7.4.2 Vegetation change

Habitat changes resulting from the encroachment of woody shrubs and small trees (vegetational succession) were identified as a threat to the species at listing. In reviewing the relevant scientific literature, much of the current threat can largely be attributed to the invasion and growth of juniper species.

- The invasion and growth of native juniper species appears to be one of the most prevalent problems in maintaining existing suitable habitat throughout a major portion of the species' range in Texas and Oklahoma. Juniper invasion has contributed to an overall afforestation of rangeland habitats throughout much of the species' breeding range in both states.

- Since listing, both Ashe juniper and redberry juniper have increased in dominance throughout the Texas range of the species; in Oklahoma, eastern redcedar has increased substantially.

- Juniper invasion into suitable habitats appears to be a function of the combined influence of fire suppression and overgrazing, and it may be further influenced by drought. At least in the eastern portion of the species' U.S. breeding range, fire appears to exert an overriding influence on the development and maintenance of breeding habitat for the species by controlling invasive juniper.

- Since listing, the increased abundance of five species of woodland birds throughout the U.S. breeding range of the black-capped vireo suggests that woody shrubs and tree cover are increasing, which would have a negative impact on black-capped vireo conservation.

7.4.3 Grazing and browsing pressures

At the time of listing, overbrowsing by domestic goats, sheep, white-tailed deer and exotic herbivores was given as a primary cause of habitat loss, particularly in the Edwards Plateau of Texas (Marshall et al. 1985). Since listing, the numbers and densities of domestic livestock have decreased throughout much of the species' U.S. breeding range, and the specific areas where livestock numbers are decreasing have generally coincided with areas where overbrowsing was most threatening to the species. However, white-tailed deer populations appear to have increased in many of the same areas (i.e., Edwards Plateau).

- Grazing *per se* is neither beneficial nor detrimental to black-capped vireo habitats. The use of grazing and browsing animals, under proper management, for enhancing rangeland habitats, is well supported in the scientific literature. However, high stocking rates combined with poor management can remove the low-growing, shrubby vegetation black-capped vireos require for breeding habitat.

- Evidence continues to suggest that extremely high stocking rates of herbivores–especially goats, white-tailed deer and exotic ungulates–can degrade black-capped vireo breeding habitat. When grazing pressure is reduced, the breeding habitat may recover under some conditions.

- Given the apparent relationship between cattle and brown-headed cowbirds, grazing by cattle may have an indirect impact on black-capped vireos by increasing the risk of brood parasitism. This relationship is highly variable and may be mitigated with livestock management and, possibly, cowbird removal.

Trends in the numbers of grazing and browsing animals.

- Goat numbers have declined throughout a major portion of the black-capped vireo's range in Texas. For example, goat numbers declined by 22.6 percent during the period 1987-2002, including decreases of 58 percent in the Southwest and Trans-Pecos region and almost 35 percent in the Edwards Plateau.

- Since listing, cattle numbers have decreased by 9.6 percent within the Texas range of the species, while increasing by about 12.5 percent in the Oklahoma portion of the range. In general, cattle densities decreased in the western portion of the species' range in Texas and increased in northeastern Texas and Oklahoma.

- Data for determining trends in grazing animals in the Mexico portion of the species' range were not available.

- Although white-tailed deer populations appear to have decreased throughout Texas since listing, deer population numbers in the Edwards Plateau appear to have increased. The resulting browsing pressure by white-tailed deer may be limiting the development of suitable habitat in many areas of that region.

- Data for estimating trends in the numbers of exotic herbivores are incomplete. However, the most recent estimates, from 1994, suggested that populations of the most numerous species (axis deer, blackbuck antelope, aoudad, fallow deer and sika deer) were increasing in the Edwards Plateau of Texas.

- The densities of domestic livestock, particularly goats, have decreased substantially in recovery regions 2 and 4 (the Edwards Plateau and Southwest and Trans-Pecos regions, respectively). However, across the Edwards Plateau, estimates of white-tailed deer densities now exceed the density estimates for all other classes of domestic livestock.

7.4.4 Brood parasitism

At the time of listing, brood parasitism by brown-headed cowbirds was widely observed as a primary factor in the low reproductive success of black-capped vireos (Marshall et al. 1985). At that time, it was thought that brown-headed cowbirds were becoming more abundant throughout the mid-continent of the U.S. and that cowbird removal was a necessary step towards black-capped vireo recovery. Important new information concerning brood parasitism and brown-headed cowbirds is summarized below.

Factors influencing abundance and parasitism rates.

- Brood parasitism rates on black-capped vireos appear to be correlated with the densities of other more conspicuous host species; this suggests that female brown-headed cowbirds may parasitize black-capped vireo nests more in areas where populations of more abundant species (e.g., northern cardinals) are denser.

- Brown-headed cowbirds often commute daily between separate feeding and breeding areas. Feeding areas are most often located with cattle; the proximity of feeding areas to breeding areas and the number of feeding sites within commuting distance are often correlated with cowbird abundance.

- At the local scale, the relationship between brown-headed cowbird abundance and rates of brood parasitism appears to be influenced by site factors such as host species assemblage, host abundance, and vegetative cover.

- At the regional scale, the threat of brood parasitism appears correlated with the regional abundance of brown-headed cowbirds.

Cowbird abundance trends.

- Throughout North America (not including Mexico), the number of brown-headed cowbirds observed along Breeding Bird Survey routes has declined by approximately 39 percent in the period 1966-2003.

- Since listing, the relative abundance of brown-headed cowbirds declined in the black-capped vireo's range in Texas, but remained stable in the species' range in Oklahoma. Over the last 10 years (1995-2004), observations of brown-headed cowbirds on BBS routes in the black-capped vireo's range in Texas have declined by 25 percent as compared to the 10-year period prior to listing (1976-1987). There was essentially no change in that comparison for Oklahoma.

Observed variability in parasitism rates.

- Observed brood parasitism rates on black-capped vireos vary across the range, with those in North-central Texas and Oklahoma being relatively higher than in other regions.

- As with other host species, the observed brood parasitism rates on black-capped vireos also can vary from year to year on any one site.

Effect of cowbird parasitism.

- The effect of cowbird parasitism on black-capped vireos is not a simple function of parasitism rates on individual nests. The effect at the population level is best measured as seasonal fecundity, which takes into account the desertion of parasitized nests, renesting attempts, remating efforts, and fledging rates. There is some evidence that high levels of brood parasitism can decrease seasonal fecundity of black-capped vireos.

- The threat posed by cowbird parasitism is proportionately greater when a species' population is declining because of other factors, such as habitat loss. In general, as a host population increases, the relative threat from brood parasitism declines.

Cowbird removal programs.

- Cowbird control programs across the black-capped vireo's range resulted in the removal of more than 235,000 cowbirds (mostly female) from 2000 through 2004.

- To date, most cowbird removal efforts in the range of the black-capped vireo are in those areas where there are relatively large black-capped vireo populations – Fort Hood

Military Reservation (Texas), Kerr WMA (Texas), Wichita Mountains WR (Oklahoma), and Fort Sill Military Reservation (Oklahoma).

- Cowbird removal can decrease local parasitism rates on black-capped vireo nests resulting in an increase in individual nest success. However, most cowbird control efforts for the black-capped vireo are combined with habitat management and restoration efforts (e.g., coordinated brush control, controlled burning, and grazing management), which confounds most attempts to determine the overall population-level benefit of cowbird removal.

7.5 Objectives of Status Review

7.5.1 Does the black-capped vireo population appear to be declining, stable, or increasing?

Detecting long-term trends in black-capped vireo populations is difficult because survey efforts across the range of the species are largely inconsistent and unequal. Several pieces of evidence should be considered in combination. These are:

- There has been a large overall increase in the total known numbers of males in breeding surveys since the time of listing.

- Most of the apparent increases in population abundance have occurred in areas of most intense survey effort (Fort Hood, Fort Sill, Wichita Mountains WR, and Kerr WMA).

- The recent preliminary results from breeding surveys in the Mexican range of the species suggest relatively large and dense breeding populations there.

- The species occurs more frequently on private lands in Texas than was known at the time of listing.[14] This increase in occurrence is at least partly a function of increased survey

---

[14] Survey results from private lands in Oklahoma are lacking.

effort – therefore it remains uncertain whether populations on private lands have actually increased.

This evidence points to two likely conclusions:

1) At the time of listing, black-capped vireos were more numerous across their breeding range than what had been documented in the listing decision because distributional information on the species was limited to a few locations at that time. We now know that the species occurs on private lands in Texas and across areas in Mexico not thought to be part of the species' breeding range when listed.

2) The overall population of black-capped vireos appears to have increased, at least in those areas receiving the most management attention and where surveys have been conducted most intensely (Fort Hood Military Reservation , Kerr WMA, Wichita Mountains WR, and Fort Sill Military Reservation). These areas contain most of the present known breeding population, yet only comprise 1 percent of the total area of rangeland in the Texas/Oklahoma range of the species. Outside of these areas, it is difficult to assess trends due to a lack of information.

7.5.2 Are threats increasing, unchanged, reduced, or eliminated?

Since listing, it appears that some threats have increased, some have decreased, and others have likely remained unchanged. And, it appears that none of the original threats to the species has been completely eliminated.

7.5.2.1 Increased threat levels

Habitat conversion and land use change. The overall loss and potential fragmentation of native rangeland caused by land use conversion and ownership changes throughout major portions of the species' breeding range, especially in the Edwards Plateau and North-central Texas regions, has likely resulted in an overall decrease in the potential habitat available for the species. While the magnitude of this threat compared to others remains unclear, it is the rate of change that is of particular concern.

Vegetation change. The widespread shift toward juniper-dominated woodlands is an issue of increasing concern for this species and others that depend upon grassland savannas and shrubland habitats. Afforestation affects the black-capped vireo throughout its U.S. breeding range in all but the more western sections of the Edwards Plateau and Southwest and Trans-Pecos regions.

Overbrowsing by white-tailed deer and exotic ungulates. While the density and abundance of domestic livestock have declined substantially in those regions of greatest concern at the time of listing, it appears that the density and abundance of white-tailed deer and exotic herbivores may have increased in many of those same areas. This is of primary concern in the Edwards Plateau of Texas. In some locations within that region, exotic ungulates may out-compete white-tailed deer.

Predation. Where cowbird removal programs are successful in reducing brood parasitism, the threat of predation from fire ants and rat snakes may become more of a threat to the species. Whereas imported fire ants have increased in distribution and abundance since the species was listed, they likely pose an increasing threat to the species.

7.5.2.2 Decreased threat levels

Overbrowsing by domestic livestock. The density and abundance of domestic livestock, particularly goats, have declined substantially in those regions where this threat was of greatest concern at the time of listing, primarily in the Edwards Plateau and Southwest and Trans-Pecos regions. Therefore, the potential for livestock overbrowsing to destroy black-capped vireo habitat is probably not as widespread as it was before listing. At the local level, however, the effects of overbrowsing by domestic livestock can be quite variable and may put local breeding populations of black-capped vireos at risk. Overall, this specific threat appears to have declined.

Brood parasitism by brown-headed cowbirds in Texas. The threat posed by brood parasitism throughout major portions of the species' range in Texas has likely lessened since the species was listed due to a combination of an apparent decrease in cowbird abundance, an apparent increase in black-capped vireo breeding populations, and circumstantial evidence of a reduction in parasitism rates at some locations due to cowbird removal. Our conclusions concerning this threat factor are confined to major portions of the Texas breeding range of the species.

7.5.2.3 Unchanged threat levels

Brood parasitism by brown-headed cowbirds in Oklahoma. The threat posed by brood parasitism throughout the species' range in Oklahoma remains essentially unchanged since the time of listing. This conclusion is largely based on the fact that the relative abundance of brown-headed cowbirds in Oklahoma appears to be unchanged. However, it is possible that brood parasitism in that region may not now be as threatening as it was thought to be at the time of listing because of the apparent increase in known black-capped vireo populations in Oklahoma and the same circumstantial evidence of beneficial cowbird removal programs at the local level as has been observed elsewhere.

7.5.3 Are there any new threats to the species?

While we did not identify any new threats to the species that were not anticipated at the time of listing, there may be unanticipated threats to the species or those that remain unexplored. Of these, nest depredation by vertebrate predators and fire ants is an obvious area for future analysis and research. This possible threat will need to be explored as new recovery strategies for the species are developed in the future. In addition, factors that might negatively affect wintering black-capped vireos in Mexico, such as development or land conversion to agriculture, need to be carefully examined.

8.0 Literature Cited

Alderson, G.W., H.L. Gibbs and S.G. Sealy. 1999. Determining the reproductive behaviour of individual brown-headed cowbirds using microsatellite DNA markers. *Animal Behaviour* 58:898-905.

Alldredge, M.W., C.C. Farquhar, D. Cimprich, and J.S. Hatfield. 2003. Survival, productivity and population growth of the black-capped vireo. *Unpublished Manuscript.*

Allendorf, F.W. 1983. Isolation, gene flow, and genetic differentiation among populations. Pages 51-65 *in* Genetics and conservation: a reference for managing wild animal and plant populations. C.M. Schonewald-Cox, S.M. Chambers, B. MacBryde, and L Thomas, editors. Benjamin Cummins, Menlo Park, California.

Allendorf, F.W. 1986. Genetic drift and the loss of alleles versus heterozygosity. Zoo Biology 5: 181-199.

Allendorf, F.W. and R.F. Leary. 1986. Heterozygosity and fitness in natural populations of animals. Pages 57-76 *in* Conservation biology: the science of scarcity and diversity. M.E. Soule, editor. Sinauer Associates, Sunderland, Massachusetts.

Ammon, E.M. and P.B. Stacey. 1997. Avian nest success in relation to past grazing regimes in a montane riparian system. *Condor* 99:7-11.

Anderson, R.C. 1982. An evolutionary model summarizing the roles of fire, climate, and grazing animals in the origin and maintenance of grasslands: an end paper. Pages 297-308 *in* Grasses and grasslands: systematics and ecology. J.R. Estes, R.J. Tyrl and J.N. Brunken, editors. University of Oklahoma Press, Norman, Oklahoma.

Ankney, C.D. 1985. Variation in weight and composition of brown-headed cowbird eggs. *Condor* 87:296-299.

Annand, E.M. and F.R. Thompson, III. 1997. Forest bird response to regeneration practices in central hardwood forests. *Journal of Wildlife Management* 61:159-171.

Ansley, R.J., W.E. Pinchak and D.N. Ueckert. 1995. Changes in redberry juniper distribution in northwest Texas. *Rangelands* 17:49-53.

Archer, S. 1989. Have southern Texas savannas been converted to woodlands in recent history? *American Naturalist* 134:545-561.

Archer, S. 1990. Development and stability of grass/woody mosaics in a subtropical savanna parkland, Texas, USA. *Journal of Biogeography* 17:453-462.

Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannahs: rates, patterns, and proximate causes. Pages 13-68 *in* Ecological implications of livestock herbivory in the West. M. Vavra, W.A. Laycock and R.D. Pieper, editors. Society for Range Management, Denver, Colorado.

Arend, J.L. 1950. Influence of fire and soil on distribution of eastern redcedar in the Ozarks. *Journal of Forestry* 48:129-130.

Askins, R.A. 1993. Population trends in grassland, shrubland, and forest birds in eastern North America. *Current Ornithology* 11:1-34.

Askins, R.A. 2000. Restoring North America's birds: lessons from landscape ecology. Yale University Press, New Haven, Connecticut.

Avise, J.C., C.F. Aquadro, and J.C. Patton. 1982. Evolutionary genetics of birds. V. Genetic distances within the Mimidae (mimic thrushes) and Vireonidae (Vireos). *Biochemical Genetics* 20: 95-104.

Bailey, J.W. 2005. Hierarchical nest-site selection and the effects of habitat characteristics on black-capped vireo nest survival. M.S. thesis, University of Missouri-Columbia, Columbia, Missouri.

Barber, D.R. 1993. Effects of alternate host densities on brown-headed cowbird parasitism rates in black-capped vireos. M.S. Thesis, University of Arkansas, Fayetteville.

Barber, D.R. and T.E. Martin. 1997. Influence of alternate host densities on brown-headed cowbird parasitism rates in black-capped vireos. *Condor* 99:595-604.

Belanger, L. and M. Picard. 1999. Cattle grazing and avian communities of the St. Lawrence River islands. *Journal of Range Management* 52:332-338.

Benson, R.H. and K.L.P. Benson. 1990. Estimated size of black-capped vireo population in northern Coahuila, Mexico. *Condor* 92:777-779.

Benson, R.H. and K.L.P. Benson. 1991. Reply to Scott and Garton. *Condor* 93:470-472.

Binford, L.C. 1989. A distributional survey of the birds of the Mexican state of Oaxaca. Ornithological Monographs No. 43. American Ornithologists' Union. Washington, D.C.

Blan, K.R. 1970. Evaluation of eastern redcedar (*Juniperus virginianus* L.) infestations in the northern Kansas flinthills. Unpublished M.S. thesis, Kansas State University, Manhattan, Kansas.

Blewett, T.J. 1986. Eastern redcedar's (*Juniperus virginiana* L.) expanded role in the prairie-forest border region. Pages 122-125 *in* The prairie: past, present, and future. Proceedings of the 9th North American Prairie Conference. G.K. Clambey and R.H. Pemble, editors. Tri-College University Center for Environmental Studies, North Dakota State University, Fargo, North Dakota.

Bocetti, C.I. 1994. Density, demography, and mating success of Kirtland's warblers in managed habitats. Ph.D. thesis, Ohio State University, Columbus, Ohio.

Bock, C.E. and J.H. Bock. 1987. Avian habitat occupancy following fire in a Montana shrubsteppe. *Prairie Naturalist* 19:153-158.

Bock, C.E. and J.H. Bock. 1988. Grassland birds in southeastern Arizona: impacts of fire, grazing, and alien vegetation. Pages 43-58 *in* Ecology and conservation of grassland birds. P.D. Goriup, editor. International Council for Bird Preservation, Publication No. 7, Cambridge, United Kingdom.

Bock, C.E., V.A. Saab, T.D. Rich and D.S. Dobkin. 1993. Effects of livestock grazing on neotropical migratory landbirds in western North America. Pages 296-309 *in* Status and management of neotropical migratory landbirds. D.M. Finch and P.W. Stangel, editors. Gen. Tech. Report RM-229. U.S. Department Agriculture, Forest Service, Rocky Mountain For. Range Exper. Station.

Bock, C.E. and B. Webb. 1984. Birds as grazing indicator species in southeastern Arizona. *Journal of Wildlife Management* 48:1045-1049.

Bogusch, E.R. 1952. Brush invasion of the Rio Grande Plains of Texas. *Texas Journal of Science* 4:85-91.

Bollinger, E.K. 1995. Successional changes and habitat selection in hayfield bird communities. *Auk* 112:720-730.

Bowman, G.B. and L.D. Harris. 1980. Effect of spatial heterogeneity on ground-nest depredation. *Journal of Wildlife Management* 44: 806-813.

Branson, F.A. 1985. Vegetation changes on western rangeland. Range Monograph 2. Society for Range Management, Denver, Colorado.

Brennan, L.A. and W.P. Kuvlesky, Jr. 2005. North American grassland birds: an unfolding conservation crisis? *Journal of Wildlife Management* 69:1-13.

Brennan, S.P. and G.D. Schnell. 2005. Relationship between bird abundances and landscape characteristics: the influence of scale. *Environmental Monitoring and Assessment* 105: 209-228.

Briskie, J.V. and S.G. Sealy. 1990. Responses of least flycatchers to experimental inter- and intraspecific brood parasitism. *Condor* 89:899-901.

Brittingham, M.C. and S.A. Temple. 1983. Have cowbirds caused forest songbirds to decline? *BioScience* 33:31-35.

Brown, B.T. 1988. Breeding ecology of a willow flycatcher population in Grand Canyon, Arizona. *Western Birds* 19:25-33.

Brown, B.T. 1993. Bell's vireo. *In* The Birds of North America, No. 35. A. Poole and F. Gill, editors. The Academy of Natural Sciences, Philadelphia, Pennsylvania, and The American Ornithologists' Union, Washington, D.C.

Brown, J.R. and S. Archer. 1987. Woody plant seed dispersal and gap formation in a North American subtropical savanna woodland: the role of domestic herbivores. *Vegetatio* 73:73-80.

Brown, J.R. and S. Archer. 1989. Woody plant invasion of grasslands: establishment of honey mesquite (*Prosopis glandulosa* var. *glandulosa*) on sites differing in herbaceous biomass and grazing history. *Oecologia* 80:19-26.

Brown, J.R. and J. Carter. 1998. Spatial and temporal patterns of exotic shrub invasion in an Australian tropical grassland. *Landscape Ecology* 13:93-102.

Bryan, K.B. and D.K. Stuart. 1990. Black-capped vireo project, 1990. Results and summary. Data from Kickapoo Caverns State Park, Devil's River State Natural Area, and Lost Maples State Park.

Burhans, D.E. and F.R. Thompson, III. 1999. Habitat patch size and nesting success of yellow-breasted chats. *Wilson Bulletin* 111:210-215.

Burnham, K.P., D.R. Anderson and J.L. Laake. 1980. Estimation of density from line transect sampling of biological populations. *Wildlife Monographs* 72:1-202.

Campbell, L. 1995. Endangered and threatened animals of Texas: their life history and management. Texas Parks and Wildlife Press, Austin, Texas.

Carter, M.D. 1986. The parasitic behavior of the bronzed cowbird in south Texas. *Condor* 88:11-25.

Chace, J.F. and A. Cruz. 1999. Influence of landscape and cowbird parasitism on the reproductive success of plumbeous vireos breeding in Colorado. *Studies in Avian Biology* 18:200-203.

Chace, J.F., C. Farmer, R. Winfree, D.R. Curson, W.E. Jensen, C.B. Goguen and S.K. Robinson. 2005. Cowbird (*Molothrus* spp.) ecology: a review of factors influencing distribution and abundance of cowbirds across spatial scales. *Ornithological Monographs* 57:45-70.

Chace, J.F., J.J. Walsh, A. Cruz, J.W. Prather and H.M. Swanson. 2003. Spatial and temporal activity patterns of the brood parasitic brown-headed cowbird at an urban/wildland interface. *Landscape and Urban Planning* 64:179-190.

Cimprich, D.A. 2002. Monitoring of the black-capped vireo during 2002 on Fort Hood, Texas. *In* Endangered species monitoring and management at Fort Hood, Texas: 2002 annual report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas, USA.

Cimprich, D. A. 2003a. The distribution of the black-capped vireo and its habitat on Fort Hood, Texas: the results of an installation-wide search. *In* Endangered species monitoring and management at Fort Hood, Texas: 2003 Annual Report. The Nature Conservancy, Fort Hood Project,Fort Hood, Texas, USA.

Cimprich, D. A. 2003b. Monitoring of the black-capped vireo during 2003 on Fort Hood, Texas. *In* Endangered species monitoring and management at Fort Hood, Texas: 2003 Annual Report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas.

Cimprich, D.A. 2004. Monitoring of the black-capped vireo during 2004 on Fort Hood, Texas. *In* Endangered species monitoring and management at Fort Hood, Texas: 2004 annual report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas, USA.

Cimprich, D. A. 2005. Monitoring of the Black-capped Vireo during 2005 on Fort Hood, Texas. *In* Endangered species monitoring and management at Fort Hood, Texas: 2005 annual report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas, USA.

Clark, K.L. and R.J. Robertson. 1981. Cowbird parasitism and evolution of antiparasite strategies in the yellow warbler. *Wilson Bulletin* 93:249- 258.

Clark, R.G., and T.D. Nudds. 1991. Habitat patch size and duck nesting success: the crucial experiments have not been performed. *Wildlife Society Bulletin* 19:534-543.

Cody, M.L. 1985. Habitat selection in birds. Academic Press, New York.

Coker, D.R., and D.E. Capen. 1995. Landscape-level habitat use by brown-headed cowbirds in Vermont. *Journal of Wildlife Management* 59: 631-637.

Cook, T.L., J.A. Koloszar, M.D. Goering and L.L. Sanchez. 1998. The spatial and temporal response of brown-headed cowbirds (*Molothrus ater*) to cattle removal. Pages 76-96 *in* The Nature Conservancy summary of 1997 research activities. Fort Hood: Texas Conservation Data Center, The Nature Conservancy.

Cooksey, M.L., J.E. Thompson. 2005. Monitoring the golden-cheeked warbler and the black-capped vireo on Camp Bullis Texas, 2005 field season report. Unpublished Report. Essex Corporation.

Coppedge, B.R., D.M. Engle, R.E. Masters and M.S. Gregory. 2004. Predicting juniper encroachment and CRP effects on avian community dynamics in southern mixed-grass prairie, USA. *Biological Conservation* 115:431-441.

Crockett, J.J. 1985. Ecology of eastern redcedar in Oklahoma. Pages 1-8 *in* Eastern redcedar in Oklahoma. R.F. Wittwer and D.M. Engle, editors. Conference proceedings, Cooperative

Extension Service, Division of Agriculture, Oklahoma State University, Stillwater, Oklahoma.

Cruz, A., T. Manolis and J.W. Wiley. 1985. The shiny cowbird: a brood parasite expanding its range in the Caribbean region. Pages 607-620 *in* neotropical ecology. P.A. Buckley, M.S. Foster, E.S. Morton, R.S. Ridgely and F.G. Buckley, editors. *Ornithological Monographs* No. 36.

Cruz, A., W. Post, J.W. Wiley, C.P. Ortega, T.K. Nakamura and J.W. Prather. 1998. Potential impacts of cowbird range expansion in Florida. Pages 313-336 *in* Parasitic birds and their hosts, studies in coevolution. S.I. Rothstein and S.K. Robinson, editors. Oxford University Press, New York, New York.

Curson, D.R. and N.E. Mathews. 2003. Reproductive costs of commuting flights in brown-headed cowbirds. *Journal of Wildlife Management* 67:520-529.

Curson, D.R., C.B. Goguen and N.E. Mathews. 2000. Long-distance commuting by brown-headed cowbirds in New Mexico. *Auk* 117: 795-799.

Davies, N.B., A.F.G. Bourke and M. de L. Brooke. 1989. Cuckoos and parasitic ants: interspecific parasitism as an evolutionary arms race. *Trends in Ecology and Evolution* 4:274-278.

Davis, S.K. and S.G. Sealy. 2000. Cowbird parasitism and nest predation in fragmented grasslands of southwestern Manitoba. Pages 220-228 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

DeCapita, M.E. 2000. Brown-headed cowbird control on Kirtland’s warbler nesting areas in Michigan, 1972-1995. Pages 333- 341 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson, and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Dobkin, D.S. 1994. Conservation and management of neotropical migrant landbirds in the Northern Rockies and Great Plains. University of Idaho Press, Moscow, Idaho.

Dobkin, D.S., A.C. Rich and W.H. Pyle. 1998. Habitat and avifaunal recovery from livestock grazing in a riparian meadow system of the northwestern Great Basin. *Conservation Biology* 12:209-221.

Donovan, T.M., F.R. Thompson III and J. Faaborg. 2000. Cowbird distribution at different scales of fragmentation: trade-offs between breeding and feeding opportunities. Pages 255-264 *In* Ecology and management of cowbirds and their hosts: studies in the conservation of

North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Dowell, B.A., J.E. Fallon, C.S. Robbins, D.K. Dawson and F.W. Fallon. 2000. Impacts of cowbird parasitism on wood thrushes and other neotropical migrants in suburban Maryland forests. Pages 244-254 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Dufault, D. 2004. Habitat occupancy by the black-capped vireo (*Vireo atricapillus*) following prescribed burns at Kerr Wildlife Management Area. M.S. Thesis, Texas State University, San Marcos, Texas.

Eckrich, G.H., T.E. Koloszar and M.D. Goering. 1999. Effective landscape management of brown-headed cowbirds at Fort Hood, Texas. *Studies in Avian Biology* 18:267-274.

Ehrlich, P.R., D.D. Dobkin and D. Wheye. 1988. The birder's handbook: a field guide to the natural history of North American birds. Simon and Schuster, New York, New York.

Engle, D.M., T.G. Bidwell and M.E. Moseley. 1995. Invasion of Oklahoma rangelands and forests by eastern redcedar and Ashe juniper. Oklahoma State University, Oklahoma Cooperative Extension Service Circ. E-947, Stillwater, Oklahoma.

Escalona, G., M. Torres, A.G. Navarro, R.Villalon, B. Hernandez and H. Benitez. 1995. Migratory birds of the cloud forests of Mexico. Pages 15-33 *in* Symposium-Workshop: Conservation of neotropical migratory birds in Mexico. M.H. Wilson and S.A. Sader, editors. Maine Agricultural and Forest Experiment Station Misc. Pub. 727.

Farquhar, C.C. and J.I. Gonzalez. 2005. Breeding habitat, distribution and population status of the black-capped vireo in northern Mexico. Project WER65 Final Report. U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

Farquhar, C.C., J.I. Gonzalez, M.A. Guerrero, O. Ballesteros. 2003. Breeding habitat, distribution, and population status of the Black-capped Vireo in Northern Mexico. Interim Report, Section 6 Grant No. E-17. Wildlife Diversity Program, Texas Parks and Wildlife Department, Austin, Texas.

Farquhar, C.C., E. Kelchlin, A. Porter, and C.H. Richart. 2005. Shrub diversity and its relationship to brood parasitism and nest depredation in the black-capped vireo: implications for management and recovery. *In preparation*.

Farquhar, C.C. and J.P. Maresh. 1996. Population biology and habitat characterization of black-capped vireos at Dolan Falls Ranch Preserve, Val Verde County, Texas. Year 1 Final Report to The Nature Conservancy of Texas.

Fazio, V.W., III. 1994. Genetic variation in an endangered Passerine, *Vireo atricapillus* Woodhouse. M.S. Thesis, Ohio University, Athens.

Fazio, V.W., III, D.B. Miles, and M.W. White. 2004. Genetic differentiation in the endangered black-capped vireo. *Condor* 106: 377-385.

Fleischer, R.C. 1985. A new technique to identify and assess the dispersion of eggs of individual brood parasites. *Behavioral Ecology & Sociobiology* 17:91-99.

Fonteyn, P.J., M.W. Stone, M.A. Yancy, J.T. Baccus and N.M. Nadkarni. 1988. Determination of community structure by fire. Pages 79-90 *in* Edwards Plateau vegetation: plant ecological studies in central Texas. B.B. Amos and F.R. Gehlbach, editors. Baylor University Press, Waco, Texas.

Foster, J. 1917. The spread of timbered areas in central Texas. *Journal of Forestry* 15:442-445.

Fowler, N. and D. Dunlap. 1986. Grassland vegetation of the eastern Edwards Plateau. *American Midland Naturalist* 115:146-155.

Franzreb, K.E. 1989a. Endangered status and strategies for protection of the least Bell's vireo (*Vireo bellii pusillus*) in California. *Western Birds* 18:43-49.

Franzreb, K.E. 1989b. Ecology and conservation of the endangered least Bell's vireo. Biological Report No. 89, U.S. Fish and Wildlife Service, Washington, D.C.

Fretwell, S.D. 1977. Is the dickcissel a threatened species? *American Birds* 31:923-932.

Friedmann, H. 1929. The cowbirds: a study in the biology of social parasitism. C.C. Thomas, Springfield, Illinois.

Friedmann, H. 1963. Host relations of the parasitic cowbirds. U.S. National Museum Bulletin 233.

Friedmann, H. and L.F. Kiff. 1985. The parasitic cowbirds and their hosts. *Proceedings of the Western Foundation of Vertebrate Zoology* 2:225-304.

Fuhlendorf, S.D. and F.E. Smeins. 1997. Long-term importance of grazing, fire and weather patterns on Edwards Plateau vegetation change. Pages 1-8 *in* 1997 Juniper symposium. C.A. Taylor, editor. Texas Agricultural Experiment Station, The Texas A & M University System. Technical Report 97-1. January 9-10, 1997, San Angelo, Texas.

Fuhlendorf, S.D., F.E. Smeins and W.E. Grant. 1996. Simulation of a fire-sensitive ecological threshold: a case study of Ashe juniper on the Edwards Plateau of Texas, USA. *Ecological Modelling* 90:245-255.

Fushille, P., and P. Ramirez. 2004. 2004 status of the black-capped vireo (*Vireo atricapilla*) on the Shield Ranch, Camp Wood, Texas. Unpublished report.

Gates, J.E. and D.R. Evans. 1998. Cowbirds breeding in the central Appalachians: spatial and temporal patterns and habitat selection. *Ecological Applications* 8:27-40.

Gehlbach, F.R. 1988. Forests and woodlands of the northeastern Balcones escarpment. Pages 57-77 *in* Edwards Plateau vegetation: plant ecological studies in central Texas. B.B. Amos and F.R. Gehlbach, editors. Baylor University Press, Waco, Texas.

Gibson, D.J., and L.C. Hulbert. 1987. Effects of fire, topography and year-to-year climatic variation on species composition in tallgrass prairie. *Vegetation* 72: 175-185.

Gilpin, M.E. and M.E. Soule. 1986. Minimum viable populations: processes of species extinction. Pages 19-34 *in* Conservation biology: the science of scarcity and diversity. M. Soule, editor. Sinauer Associates, Sunderland, Massachusetts.

Goering, M.D. 1998. Mapping the extent of the 1996 wildfire on Fort Hood using LANDSAT TM satellite imagery, color infrared photography, and GIS. Pages 111-119 *in* The Nature Conservancy. Summary of 1997 research activities. Fort Hood Project, The Nature Conservancy, Fort Hood, Texas.

Goguen, C.B. and N.E. Mathews. 1999. Review of the causes and implications of the association between cowbirds and livestock. *Studies in Avian Biology* 18:10-17.

Goguen, C.B. and N.E. Mathews. 2000. Local gradients of cowbird abundance and parasitism relative to livestock grazing in a western landscape. *Conservation Biology* 14:1862-1869.

Goguen, C.B. and N.E. Mathews. 2001. Brown-headed cowbird behavior and movements in relation to livestock grazing. *Ecological Applications* 11:1533-1544.

Goldwasser, S.D. 1981. Habitat requirements of the least Bell's vireo. Final Report, California Department of Fish and Game, Sacramento.

Goldwasser, S., D. Gaines and S.R. Wilbur. 1980. The least Bell's vireo in California: a de facto endangered race. *American Birds* 34:742-745.

Goss, N.S. 1891. History of the birds of Kansas. Museum of Natural History, University of Kansas Miscellaneous Publication No. 41.

Graber, J. 1957. A bioecological study of the black-capped vireo (*Vireo atricapilla*). PhD Dissertation. University of Oklahoma, Norman.

Graber, J.W. 1961. Distribution, habitat requirement, and life history of the black-capped vireo (*Vireo atricapilla*). *Ecological Monographs* 31:313-336.

Greaves, J.M. 1987. Nest-site tenacity of least Bell's vireo in California: a de facto endangered race. *American Birds* 34:742-745.

Greenman, T.M. 1995. Postfire characteristics of existing and potential black-capped vireo (*Vireo atricapillus*) nesting habitat on the Wichita Mountains Wildlife Refuge, Oklahoma. M.S. thesis, Oklahoma State University, Stillwater, Oklahoma.

Griffith, J.T. and J.C. Griffith. 2000. Cowbird control and the endangered least Bell's vireo: a management success story. Pages 342- 356 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Grzybowski, J.A. 1985a. Population and nesting ecology of the black-capped vireo (*Vireo atricapillus*). Part I: Population status of the black-capped vireo in Oklahoma-1985. Office of Endangered Species, U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

Grzybowski, J.A. 1985b. Final report: population and nesting ecology of the black-capped vireo (*Vireo atricapillus*). Part II: nesting ecology of the black-capped vireo. Report to U.S. Fish and Wildlife Service, Office of Endangered Species, Albuquerque, New Mexico.

Grzybowski, J.A. 1986. Interim report: population and nesting ecology of the black-capped vireo (*Vireo atricapillus*). Office of Endangered Species, U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

Grzybowksi, J.A. 1987. Performance report: population and nesting ecology of the black-capped vireo (*Vireo atricapillus*) in Oklahoma. Oklahoma Department of Wildlife Conservation Project E-1-1, Oklahoma City, Oklahoma.

Grzybowksi, J.A. 1988. Black-capped vireo investigations- 1986-1987. Part I: population and nesting ecology. Report to U.S. Fish and Wildlife Service, Office of Endangered Species, Albuquerque, New Mexico.

Grzybowski, J.A. 1989. Population and nesting ecology of the black-capped vireo (*Vireo atricapilla*) in Oklahoma. Final Report, Project E-1-3. Oklahoma Department of Wildlife and Conservation.

Grzybowski, J.A. 1990a. Ecology and management of the black-capped vireo in the Wichita Mountains, Oklahoma. Wichita Mountains National Wildlife Refuge, U.S. Fish and Wildlife Service, Indiahoma, Oklahoma.

Grzybowski, J.A. 1990b. Population and nesting ecology of the black-capped vireo- 1990. Resource Protection Division, Texas Parks and Wildlife Department, Austin, Texas.

Grzybowski, J.A. 1991. Survivorship, dispersal, and population structure of the black-capped vireo (*Vireo atricapillus*) in Oklahoma. Oklahoma Department of Conservation Performance Report E-1-6.

Grzybowski, J.A. 1995. Black-capped vireo (*Vireo atricapillus*). *In* The Birds of North America, No. 181. A. Poole and F. Gill, editors. The Academy of Natural Sciences, Philadelphia, Pennsylvania, and The American Ornithologists' Union, Washington, D.C.

Grzybowski, J.A. 2003. Status report: the black-capped vireo at Fort Sill, Oklahoma - 2002. Submitted to Fort Sill Natural Resources Division, Directorate of Public Safety.

Grzybowski, J.A. 2005. Status Report: The black-capped vireo at Fort Sill, Oklahoma - 2004. Submitted to Fort Sill Natural Resources Division, Directorate of Environmental Quality.

Grzybowski, J.A., R.B. Clapp and J.T. Marshall. 1984. Interim status report on the black-capped vireo. Office of Endangered Species, U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

Grzybowski, J.A., R.B. Clapp and J.T. Marshall. 1986. History and current status of the black-capped vireo in Oklahoma. *American Birds* 34:313-336.

Grzybowski, J.A. and V.W. Fazio III. 1991. Shiny cowbird reaches Oklahoma. *American Birds* 45:50-52.

Grzybowski, J.A. and C.M. Pease. 1999. A model of the dynamics of cowbirds and their host communities. *Auk* 116: 209-222.

Grzybowski, J.A. and C.M. Pease. 2000. Comparing the relative effects of brood parasitism and nest predation on seasonal fecundity in passerine birds. *In* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Grzybowski, J.A., D.J. Tazik and G.D. Schnell. 1994. Regional analysis of black-capped vireo breeding habitats. *Condor* 96:512-544.

Guilfoyle, M.P. 2002. Black-capped vireo and golden-cheeked warbler populations potentially impacted by USACE reservoir operations. EMRRP Technical Notes Collection (TN EMRRP-SI-28), U.S. Army Engineer Research and Development Center, Vicksburg, Mississippi.

Hahn, D.C. and R.J. O'Connor. 2002. Contrasting determinants of abundance in ancestral and colonized ranges of an invasive brood parasite. Pages 219-228 *in* Predicting species occurrences: issues of accuracy and scale. J.M. Scott, P.L. Heglund, M.L. Morrison, J.B. Haufler, M.G. Raphael, W.A. Wall and F.B. Samson, editors. Island Press, Washington, D.C.

Hahn, D.C., J.A. Sedgwick, I.S. Painter and N.J. Casna. 1999. A spatial and genetic analysis of cowbird host selection. Pages 204-217 *in* Research and management of the brown-headed cowbird in western landscapes. M.L. Morrison, L.S. Hall, S.K. Rothstein, D.C. Hahn and T. Rich, editors. *Studies in Avian Biology* No. 18.

Hall, L.S. and S.I. Rothstein. 1999. Section III: cowbird control: the efficacy of long-term control and proposed alternatives to standard control practices. *Studies in Avian Biology* 18:254-259.

Halterman, M.D., S. Allen and S.A. Laymon. 1999. Assessing the impact of brown-headed cowbird parasitism in eight national parks. *Studies in Avian Biology* 18:153-150.

Harmel, D.E. and G.W. Litton. 1981. Deer management in the Edwards Plateau of Texas. Fed. Aid in Wildl. Rest. Projects W-56-D, W-62-R, W-76-R, W-109-R, and W-14-C. PWD 7000-86. Texas Parks and Wildlife Department, Austin, Texas.

Hayden, T.J., J.D. Cornelius, H.J. Weinberg, L.A. Jette and R.H. Melton. 1999. Endangered species management plan for Fort Hood, Texas; FY99-03. Public review draft. Prepared for Army Corps US Engineers, Headquarters III Corps and Fort Hood, USACERL Technical Report 99/Draft. United States Army Construction Engineering Research Laboratories, Champaign, Illinois.

Hayden, T.J. and D.J. Tazik. 1991. Project status report: 1991 field studies of two endangered species (the black-capped vireo and the golden-cheeked warbler) and the cowbird control program on Fort Hood, Texas. Department of the Army, HQ III Corps, Fort Hood, Texas.

Hayden, T.J., D.J. Tazik, R.H. Melton and J.D. Cornelius. 2000. Cowbird control programs at Fort Hood, Texas: lessons for mitigation of cowbird parasitism on a landscape scale. Pages 357-370 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson, and S.G. Sealy, editors. The University of Texas Press, Austin, Texas.

Hedrick, P.W. and P.S. Miller. 1992. Conservation genetics: techniques and fundamentals. *Ecological Applications* 2: 30-46.

Hejl, S. J. and J.S. Young. 1999. Brown-headed cowbirds in ponderosa pine/Douglas-fir dominated-landscapes in the northern Rocky Mountains. *Studies in Avian Biology* 18:73-79.

Herkert, J.R. 1995. An analysis of Midwestern breeding bird population trends: 1966-1993. *American Midland Naturalist* 134:41-50.

Herkert, J.R., D.L. Reinking, D.A. Wiedenfeld, M. Winter, J.L. Zimmerman, W.E. Jensen, E.J. Finch, R.R. Koford, D.H. Wolfe, S.K. Sherrod, M.A. Jenkins, J. Faaborg and S.K.

Robinson. 2003. Effects of prairie fragmentation on the nest success of breeding birds in the midcontinental United States. *Conservation Biology* 17:587-594.

Hochachka, W.M., T.E. Martin, V. Artman, C.R. Smith, S.J. Hejl, D.E. Anderson, D. Curson, L. Petit, N. Mathews, T. Donovan and Others. 1999. Scale dependence in the effects of forest coverage on parasitization by brown-headed cowbirds. *Studies in Avian Biology* 18:80-88.

Holecheck, J.L., R.D. Piper and C.H. Herbel. 1989. Range management: principles and practices. Prentice-Hall, Englewood Cliffs, New Jersey.

Holmes, R.T., P.P. Marra, and T.W. Sherry. 1996. Habitat-specific demography of breeding black-throated blue warblers (*Dendroica caerulescens*): implications for population dynamics. *Journal of Animal Ecology* 65: 183-195.

Hoover, J.P. and M.C. Brittingham. 1993. Regional variation in brood parasitization of wood thrushes. *Wilson Bulletin* 105:228-238.

Howell, S.N.G. 1999. A bird-finding guide to Mexico. Cornell University Press, Ithaca, New York.

Howell, S.N.G. and S. Webb. 1995. A guide to the birds of Mexico and northern central America. Oxford University Press, New York, New York.

Humphrey, R.R. 1987. 90 years and 535 miles. Vegetation changes along the Mexican border. University of New Mexico Press, Albuquerque, New Mexico.

Hutto, R.L. 1989. Habitat distribution of migratory landbird species in western Mexico. Pages 221-239 *in* Ecology and conservation of neotropical migrant landbirds. J.M. Hagan and D.W. Johnston, editors. Smithsonian Institution Press, Washington, D.C.

Hutto, R.L. 1994. The composition and social organization of mixed-species flocks in a tropical deciduous forest in western Mexico. *Condor* 96:105-118.

James, F.C. and N.O. Wamer. 1982. Relationships between temperate forest bird communities and vegetation structure. *Ecology* 63:159-171.

James, F.C., C.E. McCulloch and D.A. Wiedenfeld. 1996. New approaches to the analysis of population trends in landbirds. *Ecology* 77:13-27.

Johnson, D.H. and L.D. Igl. 2001. Area requirements of grassland birds: a regional perspective. *Auk* 118:24-34.

Johnson, D.H. and M.D. Schwartz. 1993. The Conservation Reserve Program: habitat for grassland birds. *Great Plains Research* 3:273-295.

Johnson, R.G. and S.A. Temple. 1990. Nest predation and brood parasitism of tallgrass prairie birds. *Journal of Wildlife Management* 54:106-111.

Johnston, M.C. 1963. Past and present grasslands of southern Texas and northeastern Mexico. *Ecology* 44:456-466.

Johnson, N.K. 1995. Speciation in vireos. I. Macro-geographic patterns of allozymic variation in the *Vireo solitarus* complex in the contiguous United States. *Condor* 97: 903-919.

Johnson, N.K., R.M. Zink, and J.A. Marten. 1988. Genetic evidence for relationships in the avian family Vireonidae. *Condor* 90: 428-445.

Juarez, E.A. 2004. Habitat relationships of seven breeding bird species in the Leon River Watershed investigated at local scales. M.S. Thesis, Texas A & M University, College Station, Texas.

Juarez, E., R.N. Wilkins, T. Cummins, and S. Hejl. 2004. Wildlife component. Leon River Restoration Project: Phase I of Texas A & M University Research. Final Report to Texas Department of Agriculture.

Kauffman, J.B. and W.C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. *Journal of Range Management* 37:430-438.

Keddy-Hector, D.P. 1992. Black-capped vireo management on Texas Parks and Wildlife Department lands. Performance Report. Texas Project No. E-1-5. Job No. 3.2. Texas Parks and Wildlife Department, Austin, Texas.

Kepler, C., G. Irvine, M.E. DeCapita and J. Weinrich. 1996. The conservation management of the Kirtland's warbler *Dendroica kirtlandii*. *Bird Conservation International* 6:11-22.

Kirsch, L.M., H.F. Duebbert and A.D. Kruse. 1978. Grazing and haying effects on habitats of upland nesting birds. *Transactions of the North American Wildlife Natural Resources Conference* 43:486-497.

Kjelland, M.E., U.P. Kreuter, G.A. Clendenin, R.N. Wilkins, X.B. Wu, E.G. Afanador, and W.E. Grant. 2006 An ecological framework and empirical assessment of land ownership fragmentation in Texas. *Journal of Environmental Management*. In Press.

Knopf, F.L. 1994. Avian assemblages on altered grasslands. *Studies in Avian Biology* 15:247-257.

Knopf, F.L. and R.W. Cannon. 1982. Structural resilience of a willow riparian community to changes in grazing practices. Pages 198-207 *in* Wildlife-livestock relationships symposium. J.M. Peek and P.D. Dalke, editors. University of Idaho, Forest, Wildlife and Range Experiment Station, Moscow, Idaho.

Koford, R.R., B.S. Bowen, J.T. Lokemoen and A.D. Kruse. 2000. Cowbird parasitism in grassland and cropland in the northern Great Plains. Pages 229-235 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Koloszar, J.A. 1998. Field studies of the black-capped vireo (*Vireo atricapillus*) on Fort Hood, Texas, USA. Pages 7-27 *in* Summary of 1997 research activities. The Nature Conservancy Texas Data Center.

Koloszar, J.A. and J.S. Horne. 2000. The spatial and temporal response of brown-headed cowbirds to a reduction in cattle stocking rates- final analysis. Pages 147-177 *in* Endangered species monitoring and management at Fort Hood, Texas: 1999 annual report. Fort Hood Project, The Nature Conservancy of Texas, Fort Hood, Texas.

Kostecke, R.M., J.A. Koloszar and D.C. Dearborn. 2003. Effect of a reduction in cattle stocking rate on brown-headed cowbird activity. *Wildlife Society Bulletin* 31:1083-1091.

Kostecke, R.M., K. Ellison and S.G. Summers. 2004. Continued range expansion by bronzed cowbirds in the southwestern United States. *Southwestern Naturalist* 49:487-492.

Kostecke, R.M., S.G. Summers, G.H. Eckrich and D.A. Cimprich. 2005. Effects of brown-headed cowbird (*Molothrus ater*) removal on black-capped vireo (*Vireo atricapilla*) nest success and population growth at Fort Hood, Texas. *Ornithological Monographs* 57:28-37.

Krueper, D., J. Bart and T.D. Rich. 2003. Response of vegetation and breeding birds to the removal of cattle on the San Pedro River, Arizona (U.S.A.). *Conservation Biology* 17:607-615.

Kus, B.E. 1999. Impacts of brown-headed cowbird parasitism on productivity of the endangered least Bell's vireo. *Studies in Avian Biology* 18:160-166.

Kus, B.E., P.P. Beck and J.M. Wells. 2003. Southwestern willow flycatcher populations in California: distribution, abundance, and potential for conservation. *Studies in Avian Biology* 26:12-21.

Kus, B.E. and M.J. Whitfield. 2005. Parasitism, productivity, and population growth: response of least Bell's vireos (*Vireo bellii pusillus*) and southwestern willow flycatchers (*Empidonax traillii extimus*) to cowbird (*Molothrus* spp.) control. *Ornithological Monographs* 57:16-27.

Lacy, R.C. 1987. Loss of genetic diversity from managed populations: interacting effects of drift, mutation, selection, and population subdivision. Conservation Biology 1: 143-158.

Lande, R. and G.F. Barrowclough. 1987. Effective population size, genetic variation, and their use in population management. Pages 87-123 *in* Viable populations for conservation. M. Soule, editor. Cambridge University Press, Cambridge, United Kingdom.

Lauenroth, W.K., D.G. Milchunas, J.L. Dodd, R.H. Hart, R.K. Heitschmidt and L.R. Rittenhouse. 1994. Effects of grazing on ecosystems of the great plains. Pages 69-100 *in* Ecological implications of livestock herbivory in the West. M. Vavra, W.P. Laycock and R.D. Pieper, editors. Society of Range Management, Denver, Colorado.

Laymon, S.A. 1987. Brown-headed cowbirds in California: historical perspectives and management opportunities in riparian habitats. *Western Birds* 18:63-70.

Lesueur, H. 1945. The ecology of the vegetation of Chihuahua, Mexico, north of parallel twenty-eight. Texas University Publication 4521.

Leyva, R.I., L.A. Graham, S.A. Tweddale, and J.M. Hill. 2004. Use of LIDAR to characterize vertical vegetation distribution to identify black-capped vireo habitat in Fort Hood, Texas. *In* Endangered species monitoring and management at Fort Hood, Texas: 2004 annual report. Fort Hood Project, The Nature Conservancy of Texas, Fort Hood, Texas, USA.

Lockwood, M. 2005. White-tailed deer population trends. Performance Report. Fed. Aid in Wildl. Rest. Projects W-127-R-13. Texas Parks and Wildlife, Austin, Texas.

Lorenzana, J.C. and S.G. Sealy. 1999. A meta-analysis of the impact of parasitism by the brown-headed cowbird on its hosts. *Studies in Avian Biology* 18:241-253.

Lowther, P.E. 1993. Brown-headed cowbird (*Molothrus ater*). *In* The Birds of North America, No. 47. A. Poole and F. Gill, editors. The Academy of Natural Sciences, Philadelphia, Pennsylvania, and The American Ornithologists' Union, Washington, D.C.

Lowther, P.E. 1995. Bronzed cowbird (*Molothrus aeneus*). *In*: The Birds of North America, No. 144. A. Poole and F. Gill, editors. The Academy of Natural Sciences, Philadelphia, Pennsylvania, and American Ornithologists' Union, Washington, D.C.

Lowther, P.E. and R.F. Johnston. 1977. Influences of habitat on cowbird host selection. *Kansas Ornithological Society Bulletin* 28:36-40.

Lyons, R.K., M.K. Owens and R.V. Machen. 1998. Juniper biology and management in Texas. Texas Cooperative Extension Publication No. E-54. Texas Cooperative Extension, The Texas A&M University System, College Station, Texas.

MacArthur, R.H., J.W. MacArthur and J. Preer. 1962. On bird species diversity. II. Predictions of bird census from habitat measurements. *American Naturalist* 96:167-174.

Mack, R.N. and J.N. Thompson. 1982. Evolution in steppe with few large, hooved mammals. *American Naturalist* 119:757-773.

Magness, D.R. 2003. Breeding and wintering bird occurrence in Texas rangelands with special reference to woody plant encroachment. M.S. thesis, Texas A & M University, College Station, Texas.

Maresh, J. 2002. Census and monitoring of black-capped vireo at Garnett Property, Montague County, Texas, 2001 field season. Texas Parks and Wildlife Department

Maresh, J.P. 2003. Census and monitoring of black-capped vireo at Dobbs Mountain Ranch, Edwards County, Texas (Year Three- 2002 Field Season). Report to Dobbs Run Ranch, LLC. Texas Parks and Wildlife Department, Austin, Texas.

Maresh, J.P. 2004. Census and monitoring of black-capped vireo at Dobbs Mountain Ranch, Edwards County, Texas (Year Five- 2004 Field Season). September 2004. Report to Dobbs Run Ranch, LLC.

Maresh, J. 2004b. Census and monitoring of black-capped vireo at The Nature Conservancy of Texas' Chandler Independence Creek Preserve, Terrell County, Texas (year five - 2004 field season). Environmental Defense.

Maresh, J. 2004c. Survey of black-capped vireo, Big Bend National Park, Brewster County, Texas, Spring 2004. Environmental Defense.

Maresh, J. 2005a. Census and monitoring of black-capped vireo in Texas. Final Report: Project WER61, Grant No. E-15. Texas Parks and Wildlife Department, Austin, Texas.

Maresh, J. 2005b. Monitoring construction activities adjacent to endangered black-capped vireo habitat on Buffalo Gap Wind Energy Project, Taylor County, Texas. Turner Biological Consulting, LLC prepared for SeaWest WindPower, Inc.

Maresh, J. and G.A. Rowell. 2000. Performance Report: Project WER61, Census and monitoring of black-capped vireo in Texas. Submitted to Texas Parks and Wildlife as required by The Endangered Species Program, Grant No. E-1-12 Endangered and Threatened Species Conservation.

Maresh, J.P., G.A. Rowell, and K. O'Neal. 1999. Roadside survey for black-capped vireo on the Edwards Plateau. Final Report for U.S. Fish and Wildlife Service Endangered Species Program, Section 6, Texas Grant E-1-9, Project No. 75. Texas Parks and Wildlife Department, Austin, Texas.

Marsden, M.B. 2005. Report of bird censusing efforts for black-capped vireo (BCVI) monitoring and restoration at Cedar Ridge Preserve, 2005. Unpublished Report – U.S. Fish & Wildlife Service Arlington, Texas.

Marshall, J. T., Jr., R. B. Clapp and J. A. Grzybowski. 1984. Interim status report: *Vireo atricapillus* Woodhouse. Black-capped Vireo. Museum Section, National Museum of Natural History, Washington, D.C.

Marshall, J.T., R.B Clapp and J.A. Grzybowski. 1985. Status Report: *Vireo atricapillus* Woodhouse. Prepared for U.S. Fish and Wildlife Service, Office of Endangered Species, Albuquerque, New Mexico.

Martin, T.E. 1993. Nest predation and nest sites: new perspectives on old patterns. *Bioscience* 43:523-532.

Martin, T.E. and J.J. Roper. 1988. Nest predation and nest-site selection of a western population of Hermit Thrush. *Condor* 90:51-57.

Maurer, B.A. and M.A. Villard. 1994. Population density: geographic variation in abundance of North American birds. *National Geographic Research and Exploration* 10:306-317.

Mayfield, H. 1965. The brown-headed cowbird with old and new hosts. *Living Bird* 4:13-28.

Mayfield, H.F. 1977. Brown-headed cowbird: agent of extermination? *American Birds* 31:107-113.

Mayfield, H.F. 1978. Brood parasitism: reducing interactions between Kirtland's warblers and brown-headed cowbirds. Pages 85-91 *in* Endangered birds: management techniques for preserving threatened species. S.A. Temple, editor. University of Wisconsin Press, Madison, Wisconsin.

Mayfield, H.F. 1983. Kirtland's warbler: victim of its own rarity? *Auk* 100:974-976.

McCoy, T.D., M.R. Ryan, L.W. Burger, II and E.W. Kurzejeski. 2001. Grassland bird conservation: CP1 vs. CP2 plantings in Conservation Reserve Program fields in Missouri. *American Midland Naturalist* 145:1-17.

McGeen, D.S. 1972. Cowbird-host relationships. *Auk* 89: 360-380.

McKinney, B.R. 1998. Surveying, monitoring and managing select endangered/threatened species of concern in the Trans-Pecos ecoregion of western Texas. U.S. Fish and Wildlife Service Final Report (No. 1448-20181-97-G932). Texas Parks and Wildlife Department, Austin, Texas.

McNaughton, S.J. 1986. On plants and herbivores. *American Naturalist* 128:765-770.

McPherson, G.R. and H.A. Wright. 1990. Effects of cattle grazing and *Juniperus pinchotii* canopy cover on herb cover and production in western Texas. *American Midland Naturalist* 123:144-151.

McPherson, G.R., H.A. Wright and D.B. Wester. 1988. Patterns of shrub invasion in semiarid Texas grasslands. *American Midland Naturalist* 120:391-397.

Milchunas, D.G., O.E. Sala and W.K. Lauenroth. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. *American Naturalist* 132:87-106.

Miller, A.H. 1955. The avifauna of the Sierra del Carmen of Coahuila, Mexico. *Condor* 57:154-178.

Mills, G.S., J.B. Dunning, Jr. and J.M. Bates. 1991. The relationship between breeding bird density and vegetation volume. *Wilson Bulletin* 103:468-479.

Moore, R.T. 1938. Unusual birds and extensions of ranges in Sonora, Sinaloa, and Chihuahua, Mexico. *Condor* 40:23-28.

Morris, D. L. and F. R. Thompson, III. 1998. Effects of habitat and invertebrate density on abundance and foraging behavior of brown-headed cowbirds. *Auk* 115:376-385.

Morse, S.F. and S.K. Robinson. 1999. Nesting success of a neotropical migrant in a multiple-use, forested landscape. *Conservation Biology* 13:327-337.

Muller, C.H. 1947. Vegetation and climate of Coahuila, Mexico. *Madrono* 9:1-32.

Murphy, M.T. 2003. Avian population trends within the evolving agricultural landscape of eastern and central United States. *Auk* 120:20-34.

Noa, L. 2005. Demographic differences of black-capped vireos (*Vireo atricapillus*) in two habitat types in central Texas. M.S. Thesis, University of Vermont, Burlington.

Nolan, V., Jr. 1978. The ecology and behavior of the prairie warbler, *Dendroica discolor*. Ornithological Monographs No. 26, American Ornithologists's Union, Washington, D.C.

Oberholser, H.C. 1974. The bird life of Texas. Volume 2. Edgar Kincaid, editor. University of Texas Press, Austin, Texas.

Ortega, C.P. 1998. Cowbirds and other brood parasites. University of Arizona Press, Tucson, Arizona.

Ortega, C.P., J.F. Chace and B.D. Peer. 2005. Research directions and cowbird (*Molothrus* spp.) management. *Ornithological Monographs* 57:1-5.

Ortega, C.P. and J.C. Ortega. 1999. Brown-headed cowbird (*Molothrus ater*) parasitism on warbling vireos (*Vireo gilvus*) in southwest Colorado. *Auk* 120:759-764.

Parysow, P. and D.J. Tazik. 2002. Assessing the effect of estimation error on population viability analysis: an example using the black-capped vireo. *Ecological Modeling* 155: 217-229.

Pavlacky, D.C., Jr. and S.H. Anderson. 2001. Habitat preferences of pinyon-juniper specialists near the limit of their geographic range. *Condor* 103:322-331.

Pease, C.M and J.A. Grzybowski. 1995. Assessing the consequences of brood parasitism and nest predation on seasonal fecundity in passerine birds. *Auk* 112:343-363.

Peden, D.G., G.M. Van Dyne, R.W. Rice and R.M. Hanson. 1974. The trophic ecology of *Bison bison* L. on shortgrass plains. *Journal of Applied Ecology* 11:489-497.

Penfound, W.T. 1968. Influence of a wildfire in the Wichita Mountains Wildlife Refuge, Oklahoma. *Ecology* 49:1003-1006.

Peterjohn, B.G. and J.R. Sauer. 1999. Population status of North American grassland birds from the North American Breeding Bird Survey, 1966-1996. *Studies in Avian Biology* 19:27-44.

Peterjohn, B.G., J.R. Sauer and S. Schwartz. 2000. Temporal and geographic patterns in population trends of brown-headed cowbirds. Pages 21-34 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Petit, L.J., and D.R. Petit. 1996. Factors governing habitat selection by prothonotary warblers: field tests of the Fretwell-Lucas models. *Ecological Monographs* 66: 367-388.

Petit, L.J. and D.R. Petit. 2000. Brown-headed cowbird parasitism of migratory birds: effects of forest area and surrounding landscape. Pages 265-270 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Phillips, J.C. 1911. A year's collecting in the state of Tamaulipas, Mexico. *Auk* 28:67-89.

Pinkston, J., J. Maresh, and N. Wright. 2002. Population monitoring for the black-capped vireo (*Vireo atricapillus*) at Fossil Rim Wildlife Center, Dinosaur Valley State Park and adjacent private property in Somervell County, Texas (2001 Field Season). Final Report. Texas Parks and Wildlife Department.

Platts, W.S. 1991. Livestock grazing. Influences of forest and rangeland management on salmonid fishes and their habitats. *American Fisheries Society Special Publication* 19:389-423.

Post, W., A. Cruz and D.B. McNair. 1993. The North American invasion pattern of the shiny cowbird. *Journal of Field Ornithology* 64:32-41.

Pyle, P. 1997. Identification guide to North American birds. Part I. Columbidae to Ploceidae. Slate Creek Press, Bolinas, California.

Ratzlaff, A. 1987. Endangered and threatened wildlife and plants: determination of the black-capped vireo to be an endangered species. *Federal Register* 52:37420-37423.

Renardo, J. 1886. Notes on some birds of the United States which occur in the Mexican fauna. *Ornithologist and Oologist* 11:132-133.

Ridgway, R. 1902. The birds of North and Middle America. *Bulletin of the United States National Museum* 50:1-834.

Riley, T.Z., C.A. Davis, M. Ortiz and M.J. Wisdom. 1992. Vegetative characteristics of successful and unsuccessful nests of lesser prairie chickens. *Journal of Wildlife Management* 56:383-387.

Risser, P.G., E.C. Birney, H.D. Blocker, S.W. May, W.J. Parton and J.A. Wiens. 1981. The true prairie ecosystem. Hutchinson Ross Publishing Company, Stroudsburg, Pennsylvania.

Robbins, C.S., J.R. Sauer, R.S. Greenberg and S. Droege. 1989. Population declines in North American birds that migrate to the Neotropics. *Proceedings of the National Academy of Sciences USA* 86:7658-7662.

Robinson, S.K., J.A. Grzybowski, S.I. Rothstein, M.C. Brittingham, L.J. Petit and F.R. Thompson. 1993. Management implications of cowbird parasitism on neotropical migrant songbirds. Pages 93-102 *in* Status and management of neotropical migratory birds. D.M. Finch and P.W. Stangel, editors. United States Department of Agriculture, Forest Service General Technical Report RM-229, Rocky Mountain Forest Range Experiment Station, Fort Collins, Colorado.

Robinson, S.K., J.P. Hoover and J.R. Herbert. 2000. Cowbird parasitism in a fragmented landscape: effects of tract size, habitat, and abundance of cowbirds and hosts. Pages 280-297 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson, and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Robinson, S.K., S.I. Rothstein, M.C. Brittingham, L.J. Petit and J.A. Grzybowski. 1995. Ecology and behavior of cowbirds and their impact on host populations. Pages 428-460 *in* T.E. Martin and D.M. Finch, editors. Ecology and management of neotropical migratory birds. Oxford University Press, New York, New York.

Robinson, S.K. and D.S. Wilcove. 1994. Forest fragmentation in the temperate zone and its effects on migratory songbirds. *Bird Conservation International* 4:233-249.

Rockwell, R.F. and G.F. Barrowclough. 1987. Gene flow and the structure of populations. Pages 223-255 *in* Avian genetics. A population and ecological approach. F. Cooke and P.A. Buckley, editors. Academic Press, London, UK.

Rodewald, A.D. and A.C. Vitz. 2005. Edge- and area-sensitivity of shrubland birds. *Journal of Wildlife Management* 69:681-688.

Rohwer, S., S.D. Fretwell and D.M. Niles. 1980. Delayed maturation in passerine plumages and the deceptive acquisition of resources. *American Naturalist* 115:400-437.

Rollins, D. 1985. Controlling eastern redcedar with prescribed fire. Pages 71-83 *in* Eastern redcedar in Oklahoma. R.F. Wittwer and D.M. Engle, editors. Conference proceedings, Cooperate Extension Service, Division of Agriculture, Oklahoma State University, Stillwater, Oklahoma.

Rotenberry, J.T. and J.A. Wiens. 1980. Habitat structure, patchiness and avian communities in North American steppe vegetation: a multivariate analysis. *Ecology* 61:1228-1250.

Roth, R.R., M.S. Johnson and T.J. Underwood. 1996. Wood thrush (*Hylocichla mustelina*). *In*: The Birds of North America, No. 246 (A. Poole and F. Gill, editors). The Academy of Natural Sciences, Philadelphia, Pennsylvania, and American Ornithologists' Union, Washington, D.C.

Rothstein, S.I. 1975. An experimental and teleonic investigation of avian brood parasitism. *Condor* 77:250-271.

Rothstein, S.I. 1976. Cowbird parasitism of the cedar waxwing and its evolutionary implications. *Auk* 93:498-509.

Rothstein, S.I. 1977. Cowbird parasitism and egg recognition of the northern oriole. *Wilson Bulletin* 89:21-32.

Rothstein, S.I. 1994. The cowbird's invasion of the Far West: History, causes and consequences experienced by host species. *Studies in Avian Biology* 15:301-315.

Rothstein, S.I. 2004. Brown-headed cowbird: Villain or scapegoat? *Birding* 36:374-384.

Rothstein, S.I., B.E. Kus, M.J. Whitfield and S.J. Sferra. 2003. Recommendations for cowbird management in recovery efforts for the southwestern willow flycatcher. *Studies in Avian Biology* 26:157-167.

Rothstein, S.I. and B.D. Peer. 2005. Conservation solutions for threatened and endangered cowbird (*Molothrus* spp.) hosts: separating fact from fiction. *Ornithological Monographs* 57:98-114.

Rothstein, S.I., and S.K. Robinson. 1998. Parasitic birds and their hosts: studies on coevolution. Oxford University Press, New York City, New York.

Rothstein, S.I., J. Verner and E. Stevens. 1984. Radio-tracking confirms a unique diurnal pattern of spatial occurrence in the parasitic brown-headed cowbird. *Ecology* 65:77-88.

Rothstein, S.I., D.A. Yokel and R.C. Fleischer. 1986. Social dominance, mating and spacing systems, female fecundity, and vocal dialects in captive and free ranging brown-headed cowbirds. Pages 127-185 *in* Current Ornithology. R.F. Johnston, editor. Plenum, New York, New York.

Russell, S.M. and G. Morrison.1996. The Birds of Sonora. University of Arizona Press, Tucson, Arizona.

Rust, S.P. and D.J. Tazik. 1990. 1990 field season report: endangered bird survey on Camp Bullis Training Reservation, Fort Sam Houston, Texas.

Ryder, R.A. 1980. Effects of grazing on bird habitats. Pages 51-66 *in* Management of western forests and grasslands for nongame birds. R.M. DeGraff and N.G. Tilghman, editors. U.S. Forest Service, Intermountain Research Station Resource Paper INT-86.

Saab, V.A., C.E. Bock, T.D. Rich and D.S. Dobkin. 1995. Livestock grazing effects in western North America. Pages 311-353 *in* Ecology and management of neotropical migratory birds: a synthesis and review of critical issues. T.E. Martin and D.M. Finch, editors. Oxford University Press, New York, New York.

Saab, C.A. and M.L. Morrison. 1999. Managing riparian vegetation to control cowbirds. *Studies in Avian Biology* 18:18-22.

Sanders, J. 2005. Relationships among landowner and land ownership characteristics and participation in conservation programs in Central Texas. M.S. Thesis. Texas A&M University, College Station, Texas.

Sansom, A. 1995. Introduction. P. iv. *In* L. Campbell. Endangered and threatened animals of Texas. Texas Parks and Wildlife Press, Austin, Texas.

Sauer, C.O. 1950. Grassland, climax, fire, and man. *Journal of Range Management* 3:16-22.

Sauer, J.R. and S. Droege. 1992. Geographic patterns in population trends of neotropical migrants in North America. Pages 26-42 *in* Ecology and conservation of neotropical migrant landbirds. J.M. Hagan and D.W. Johnston, editors. Smithsonian Institution Press, Washington, D.C.

Sauer, J.R., J.E. Hines and J. Fallon. 2003. The North American Breeding Bird Survey, results and analysis 1966-2002, version 2003.1. United States Geological Survey, Patuxent Wildlife Research Center, Laurel, Maryland.

Sauer, J.R., J.E. Hines and J. Fallon. 2004. The North American Breeding Bird Survey, results and analysis 1966-2003, version 2004.1. United States Geological Survey, Patuxent Wildlife Research Center, Laurel, Maryland.

Sauer, J.R., J.E. Hines and J. Fallon. 2005. The North American Breeding Bird Survey, results and analysis 1966-2004, version 2005.2. United States Geological Survey, Patuxent Wildlife Research Center, Laurel, Maryland.

Sauer, J.R., B.J. Peterjohn, S. Schwartz and J.E. Hines. 1996. The North American Breeding Bird Survey results and analysis. Version 95.1. Patuxent Wildlife Research Center, Laurel, Maryland.

Scanlan, J.C. and S. Archer. 1991. Simulated dynamics of succession in a North American subtropical *Prosopis* savanna. *Journal of Vegetation Science* 2:625-634.

Schlesinger, W.H., J.F. Reynolds, G.L. Cunningham, L.F. Huenneke, W.M. Jarrell, R.A. Virginia and W.G. Whitford. 1990. Biological feedbacks in global desertification. *Science* 247:1043-1048.

Schwartz, C.C. and J.E. Ellis. 1981. Feeding ecology and niche separation in some native and domestic ungulates on the shortgrass prairie. *Journal of Applied Ecology* 18:343-353.

Schultz, T.T. and W.C. Leininger. 1990. Differences in riparian vegetation structure between grazed areas and exclosures. *Journal of Range Management* 43:295-299.

Scott, D.M. 1977. Cowbird parasitism on the gray catbird at London, Ontario. *Auk* 94:18.

Scott, D.M. and C.D. Ankney. 1983. The laying cycle of brown-headed cowbirds: passerine chickens? *Auk* 100:583-592.

Scott, J.M. and E.O. Garton. 1991. Population estimates of the black-capped vireo. *Condor* 93:469-470.

Scott, M.L., S.K. Skagen and M.F. Merigliano. 2003. Relating geomorphic change and grazing to avian communities in riparian forests. *Conservation Biology* 17:284-296.

Sealy, S.G. 1992. Removal of yellow warbler eggs in association with cowbird parasitism. *Condor* 94:40-54.

Sedgwick, J.A. and W.M. Iko. 1999. Costs of brown-headed cowbird parasitism to willow flycatchers. *Studies in Avian Biology* 18:167-181.

Sedgwick, J.A. and F.L. Knopf. 1988. A high incidence of brown-headed cowbird parasitism on willow flycatchers. *Condor* 90:253-256.

Sennett, G.B. 1878. Notes on the ornithology of the lower Rio Grande of Texas made during the season of 1877. *Bulletin of the United States Geological Survey* 4:1-66.

Severson, K.E. and P.J. Urness. 1994. Livestock grazing: a tool to improve wildlife habitat. Pages 232-249 *in* Ecological implications of livestock herbivory in the West. M. Vavra, W.A. Laycock and R.D. Pieper, editors. Society for Range Management, Denver, Colorado.

Sexton, C.W. 2002. List of black-capped vireo populations on Balcones Canyonlands NWR. Unpublished manuscript.

Sexton, C.W. 2005. Black-capped vireo populations on Balcones Canyonlands National Wildlife Refuge. Unpublished data. U.S. Fish and Wildlife Service.

Shackford, J.S. 2004. Annual Report 2004: Black-capped vireo at Lake Stanley Drapper in central Oklahoma in 2004. Submitted to U.S. Fish and Wildlife Service, Albuquerque, New Mexico, USA.

Shaw, D.M., D.J. Tazik, J.M. Coffey and S.F. Atkinson. 1989. Status report: the black-capped vireo on Camp Bullis, Fort Sam Houston, Texas. Report submitted to Fort Worth District Corps of Engineers. Center for Remote Sensing and Landuse Analysis, University of North Texas, Denton, Texas.

Shull, A.M. 1986. Endangered and threatened wildlife and plants: proposal to determine the black-capped vireo to be an endangered species. Federal Register 51: 44808-44812.

Slatkin, M. 1985. Gene flow in natural populations. *Annual Review of Ecology and Systematics* 16: 393-430.

Smeins, F.E., S. Fuhlendorf, and C. Taylor, Jr. 1997. Environmental and landuse changes: a long term perspective. Pages 1-3 through 1-21 in 1997 Juniper symposium. C.A. Taylor, editor. Texas Agricultural Experiment Station, The Texas A&M University System. Technical Report 97-1. January 9-10, 1997, San Angelo, Texas.

Smeins, F.E. and L.B. Merrill. 1988. Long-term change in a semiarid grassland. Pages 101-114 *in* Edwards Plateau vegetation: plant ecological studies in central Texas. B.B. Amos and F.R. Gehlbach, editors. Baylor University Press, Waco, Texas.

Smith, J.N.M. 1981. Cowbird parasitism, host fitness, and age of the host female in an island song sparrow population. *Condor* 83:152-161.

Smith, J.N.M. 1999. The basis for cowbird management: host selection, impacts of hosts, and criteria for taking management action. *Studies in Avian Biology* 18:104-108.

Smith, J.N.M. and I.H. Myers-Smith. 1998. Spatial variation in parasitization of song sparrows by brown-headed cowbirds. Pages 296-312 *in* Parasitic birds and their hosts. S.I. Rothstein and S.K. Robinson, editors. Oxford University Press, New York City.

Snook, E.C. 1985. Distribution of eastern redcedar on Oklahoma rangelands. Pages 45-52 *in* Eastern redcedar in Oklahoma. R.F. Wittwer and D.M. Engle, editors. Conference proceedings, Cooperate Extension Service, Division of Agriculture, Oklahoma State University, Stillwater, Oklahoma.

Sparkman, L.J. 1996. Effects of moderate human intrusion on singing behavior of black-capped vireos (*Vireo atricapillus*), Hill Country State Natural Area, Texas. M.S. thesis, Southwest Texas State University, San Marcos, Texas.

Stake, M.M., and P.M. Cavanagh. 2001. Removal of host nestlings and fecal sacs by Brown-headed Cowbirds. *Wilson Bulletin* 113:456-459.

Stake, M.M. and D.A. Cimprich. 2003. Using video to monitor predation at black-capped vireo nests. *Condor* 105:348-357.

Stangel, P.W., M.R. Lennartz, and M.H. Smith. 1992. Genetic variation and population structure of red-cockaded woodpeckers. *Conservation Biology* 6: 283-290.

Stebbins, G.L. 1981. Coevolution of grasses and herbivores. *Ann. Missouri Botanical Garden* 68:75-86.

Steinauer, E.M. and T.B. Bragg. 1987. Ponderosa pine (*Pinus ponderosa*) invasion of Nebraska sandhills prairie. *American Midland Naturalist* 118:358-365.

Stewardship Services. 1988. Project summary report: black-capped vireo survey for Cedars Development Corporation.

Stribley, J.M. and J.B. Haufler. 1999. Landscape effects on cowbird occurrences in Michigan: implications to research needs in forests of the inland West. *Studies in Avian Biology* 18:68-72.

Stritzke, J.F. and T.G. Bidwell. 1990. Eastern redcedar and its control. OSU Extension Facts No. 2850. Cooperative Extension Service, Division of Agriculture, Oklahoma State University, Stillwater, Oklahoma.

Summers, S.G., G.H. Eckrich and W.C. Holimon. 2000. Brown-headed cowbird control program on Fort Hood, Texas. *In* Endangered species monitoring and management at Fort Hood, Texas: 1999 annual report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas, USA.

Summers, S.G. and G.L. Norman. 2002. Brown-headed cowbird removal at Fort Hood, Texas, 2001-2002. *In* Endangered species monitoring and management at Fort Hood, Texas: 2002 annual report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas, USA.

Summers, S.G. and G.L. Norman. 2003. Brown-headed cowbird removal at Fort Hood, Texas, 2002-2003. *In* Endangered species monitoring and management at Fort Hood, Texas: 2003 annual report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas, USA.

Summers, S.G. and G.L. Norman. 2004. Brown-headed cowbird removal at Fort Hood, Texas, 2003-2004. *In* Endangered species monitoring and management at Fort Hood, Texas: 2004 annual report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas, USA.

Summers, S.G. and J.L. Sterling. 2001. Brown-headed cowbird control program on Fort Hood, Texas, 2001. *In* Endangered species monitoring and management at Fort Hood, Texas: 2001 annual report. The Nature Conservancy, Fort Hood Project, Fort Hood, Texas, USA.

Taylor, D.M. 1986. Effects of cattle grazing on passerine birds nesting in riparian habitat. *Journal of Range Management* 39:254-258.

Tazik, D.J. 1988. The black-capped vireo on the lands of Fort Hood, Texas: a summary of research conducted during 1987 and 1988. September 1988. U.S. Army Construction Engineering Research Laboratories, Champaign, Illinois.

Tazik, D.J. 1991a. Biological assessment of the effects of military associated activities on endangered species at Fort Hood, Texas. Report submitted to HQ III Corps and Fort Hood, Directorate of Engineering and Housing, Fort Hood, Texas.

Tazik, D.J. 1991b. Proactive management of an endangered species on army lands: the black-capped vireo on the lands of Fort Hood, Texas. Ph.D. dissertation, University of Illinois, Urbana-Champaign, Illinois.

Tazik, D.J. and J.D. Cornelius. 1989. The black-capped vireo on the lands of Fort Hood, Texas. Preliminary Status Report, Directorate of Engineering and Housing, Fort Hood, Texas.

Tazik, D.J. and J.D. Cornelius. 1993. Status of the black-capped vireo at Fort Hood, Texas. Vol. 3, Population and nesting ecology. U.S. Army Construction Engineering Research Laboratories Technical Report EN-94/01/ADA 277544, Champaign, Illinois.

Tazik, D.J., J.D. Cornelius and C. Abrahamson. 1993. The black-capped vireo on the lands of Fort Hood, Texas, Volume I: distribution and abundance. TR EN-94/01. USACERL.

Tazik, D.J., J.A. Grzybowski, and J.D. Cornelius. 1989. Habitat of the endangered black-capped vireo on the lands of Fort Hood, Texas. Interim Report, U.S. Army Construction Engineering Research Laboratory, Champaign, Illinois.

Tazik, D.J., J.A. Grzybowski and J.D. Cornelius. 1990. The black-capped vireo on the lands of Fort Hood, Texas. Part II: Habitat. Draft Technical Report. U.S. Army Construction Engineering Research Laboratory, Champaign, Illinois.

Terborgh, J.W. 1989. Where have all the birds gone? Princeton University Press, Princeton, New Jersey.

Tewksbury, J.J., A.E. Black, N. Nur, V.A. Saab, B.D. Logan and D.S. Dobkin. 2002. Effects of anthropogenic fragmentation and livestock grazing on western riparian bird communities. *Studies in Avian Biology* 25:158-202.

Tewksbury, J.J., T.E. Martin, S.J. Hejl, T.S. Redman and F.J. Wheeler. 1999. Cowbirds in a western valley: effects of landscape structure, vegetation, and host density. *Studies in Avian Biology* 18:23-33.

Texas Parks and Wildlife Department. 2005. Texas Wildlife Diversity Database. Austin, Texas. (Data retrieved 14 September 2005)

Tharp, B. 1926. Structure of Texas vegetation east of the 98th meridian. University of Texas Bulletin 2606. Austin, Texas.

Thompson, F.R. 1994. Temporal and spatial patterns of breeding brown-headed cowbirds in the Midwestern United States. *Auk* 111:979-990.

Thompson, F.R., III, S.K. Robinson, T.M. Donovan, J. Faaborg, D.R. Whitehead and D.R. Larsen. 2000. Biogeographic, landscape, and local factors affecting cowbird abundance and host parasitism levels. Pages 271-279 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson, and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Thurow, A. and T. Thurow. 1997. Juniper management in the Edwards Plateau: policy issues and options. Pages 13-25 *in* 1997 Juniper symposium. C.A. Taylor, editor. Texas Agricultural Experiment Station, The Texas A & M University System. Technical Report 97-1. January 9-10, 1997, San Angelo, Texas.

Thurow, T., A. Thurow, C. Taylor, R. Conner and M. Garriga. 1997. Environmental and economic tradeoffs associated with vegetation management on the Edwards Plateau. Chapter 2 *in* 1997 Juniper symposium. C.A. Taylor, editor. Texas Agricultural Experiment Station, The Texas A & M University System. Technical Report 97-1. January 9-10, 1997, San Angelo, Texas.

Tordoff, H.B. 1956. Checklist of the birds of Kansas. Univ. Kansas Mus. Nat. Hist. Publ. 8: 307-359.

Trame, A., S.J. Harper and J. Westervelt. 1998. Management of cowbird traps on the landscape: an individual-based modeling approach for Fort Hood, Texas. U.S. Army Corps of Engineers, Construction Engineering Research Laboratories Technical Report 98/121.

Traweek, M.S. 1995. Statewide census of exotic big game animals. Fed. Aid in Wildl. Rest. Report W-127-R-3, Project No. 21. Texas Parks and Wildlife Department, Austin, Texas.

Traweek, M.S. and R. Welch. 1992. Exotics in Texas. PWD BK W7000-206 (5/92). Texas Parks and Wildlife Department, Austin, Texas.

U.S. Fish and Wildlife Service. 1991. Black-capped vireo (*Vireo atricapillus*) recovery plan. Austin, Texas.

U.S. Fish and Wildlife Service. 1996. Black-capped vireo population and habitat viability assessment report. C. Beardmore, J. Hatfield and J. Lewis, editors. Report of a September 18-21, 1995 workshop in partial fulfillment of U.S. National Biological Service Grant No. 80333-1423. Austin, Texas. Submitted to U.S. Department of Agriculture, Temple, Texas.

U.S. Fish and Wildlife Service. 2002. Southwestern willow flycatcher recovery plan. Albuquerque, New Mexico.

U.S. Fish and Wildlife Service. 2004. Biological Opinion. Implementation of 2002 Farm Bill conservation programs and effects on federally listed species (consultation with Section 7 of the Endangered Species Act). Report No. 2-12-05-F-021. U.S. Department of Interior, Fish and Wildlife Service, Arlington, Texas.

Ueckert, D. 1997. Juniper control and management. Pages 23-34 *in* 1997 Juniper symposium. C.A. Taylor, editor. Texas Agricultural Experiment Station, The Texas A&M University System. Technical Report 97-1. January 9-10, 1997, San Angelo, Texas.

Unitt, P. 1987. *Empidonax trailii extimus*: an endangered subspecies. *Western Birds* 18:137-162.

Van Hoose, S.G. 1955. Distributional and breeding records of some birds from Coahuila. *Wilson Bulletin* 67:302-303.

Vavra, M., W.P. Laycock, R.D. Pieper. 1994. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, Colorado.

Vickery, P.D. P.L. Tubaro, J.M.C. da Silva, B.G. Peterjohn, J.R. Herkert and R.B. Cavalcanti. 1999. Conservation of grassland birds in the western hemisphere. Pages 2-26 *in* Ecology and conservation of grassland birds of the western hemisphere. P.D. Vickery and J.R. Herkert, editors. *Studies in Avian Biology* 19:2-26.

Villard, M.A., G. Merriam, and B.A. Maurer. 1995. Dynamics in subdivided populations of neotropical migratory birds in a fragmented temperate forest. *Ecology* 76: 27-40.

Ward, M.P. and S. Schlossberg. 2004. Conspecific attraction and the conservation of territorial songbirds. Conservation Biology. 18(2):519-525.

Wauer, R.H. and J.D. Ligon. 1977. Distributional relations of breeding avifauna of four southwestern mountain ranges. Pages 567-578 in Transactions of the symposium on the biological resources of the Chihuahua Desert Region, United States and Mexico. R.H. Wauer and D.H. Riskind, editors. U.S. Department of Interior, National Park Service.

Webster, F.S. 1962. The black-capped vireo in central Texas. *Audubon Field Notes* 16:414-417.

Whitfield, M.J. and M.K. Sogge. 1999. Range-wide impact of brown-headed cowbird parasitism on the southwestern willow flycatcher (*Empidonax traillii extimus*). *Studies in Avian Biology* 18:182-190.

Whitfield, M.J., K.M. Enos and S.P. Rowe. 1999. Is brown-headed cowbird trapping effective for managing populations of the endangered southwestern willow flycatcher? Pages 260-266 *in* Research and management of the brown-headed cowbird in western landscapes. M.L. Morrison, L.S. Hall, S.K. Robinson, S.I. Rothstein, D.C. Hahn and T. Rich, editors. *Studies in Avian Biology* No. 18.

Wiedenfeld, D.A. 2000. Cowbird population changes and their relationship to changes in some host species. Pages 35-46 *in* Ecology and management of cowbirds and their hosts: studies in the conservation of North American passerine birds. J.N.M. Smith, T.L. Cook, S.I. Rothstein, S.K. Robinson and S.G. Sealy, editors. University of Texas Press, Austin, Texas.

Wiley, J.W. 1988. Host selection by the Shiny Cowbird. *Condor* 90:289-303.

Wilkins, R.N., A. Hays, D. Kubenka, D. Steinbach, W. Grant, E. Gonzalez, M. Kjelland, and J. Shackelford. 2003. Texas Rural Lands: Trends and Conservation Implications for the 21st Century. Final Summary Report of the Texas A&M Rural Land Fragmentation Project. Texas Cooperative Extension Publication B6134.

Wilkins , R.N. , S. Hejl, D. Magness, and T. Bedford. 2002. Wildlife Response to Brush Management Scenarios. Pp 110-133 In Ecosystem and Wildlife Management Implications of Brush Management Systems Designed to Improve Water Yield. Final Report. U.S. Army Corps of Engineers. http://landinfo.tamu.edu/projects/Brush/brush.htm

Willson, M.F. 1974. Avian community organization and habitat structure. *Ecology* 55:1017-1029.

Woodward, A.A., A.D. Fink and F.R. Thompson, III. 2001. Edge effects and ecological traps: effects on shrubland birds in Missouri. *Journal of Wildlife Management* 65:668-675.

Woolfenden, B.E., H. L. Gibbs, S.G. Sealy and D.G. McMaster. 2003. Host use and fecundity of individual female brown-headed cowbirds. *Animal Behaviour* 66:95-106.

Wray, T., II and R.C. Whitmore. 1979. Effects of vegetation on nesting success of vesper sparrows. *Auk* 96:802-805.

Wright, S. 1931. Evolution in Mendelian populations. *Genetics* 16: 97-169.

Young, J. S. and R.L. Hutto. 1999. Habitat and landscape features affecting cowbird distribution in the northern Rockies. *Studies in Avian Biology* 18:41-51.

Zanette, L., E. MacDougall-Shakleton, M.Clinchy and J.N.M. Smith. 2005. Brown-headed cowbirds skew host offspring sex ratios. *Ecology* 86:815-820.

9.0 Appendices

APPENDIX A. Current known population of black-capped vireos arranged by county and recovery regions.

REGION 1- North-central Texas

County	Specific Location	Current Population[1]	Current Source by Specific Location	Texas Wildlife Diversity Database[2]	1996 Population[3]
Bell/Coryell[4]		**1914**			
	Fort Hood		1,847 males (Cimprich 2003b)	594 males (1999)	
Bell					**150**
Coryell					**150**
	Private land		16 individuals (Butcher pers. comm. 2005)		
	Private land		50 males (Juarez et al. 2004)		
	Private land		1 male (Maresh and Rowell 2000)		
Burnet		**88**		39 males (2001)	**47**
	Balcones Canyonlands NWR		57 territories (Sexton 2002)		
	Private Land		8 territories (Sexton 2005)		
	LCRA Canyon of the Eagles		23 males (Pavlas pers. comm. 2004[6])		
Travis		**43**		5 males (2001)	**60**
	Balcones Canyonlands NWR		31 territories (Sexton 2002)		
	Balcones Canyonlands Preserve (BCP)		12 pairs (Ramirez pers. comm. 2004[5])		
Williamson		**14**		5 males (2001)	**13**
	Balcones Canyonlands NWR		14 territories (Sexton 2002)		

REGION 1- North-central Texas (continued)

County	Specific Location	Current	Current Source by Specific Location	Texas Wildlife	1996

		Population[1]		**Diversity Database[2]**	**Population[3]**
Somervell		**20**		16 males (2003)	**3**
	Private land		15 males (Maresh 2005a)		
	Private land		2 pairs (Pinkston et al. 2002)		
	Fossil Rim Wildlife Center		3 males (Pinkston et al. 2002)		
Bosque		**1**		1 male (1988)	**1**
	County Road 2130		1 pair (Maresh and Rowell 2000)		
Coleman		**6**		6 males (1999)	**unconfirmed**
	County Road 419		6 males (Maresh and Rowell 2000)		
Dallas	Cedar Ridge Preserve	**1**	1 male (Marsden 2005)	2 males (1993)	**"extirpated"**
Erath		**16**		16 males (2003)	**1**
Montague		**2**		1 male (2001)	**no record**
	Private land		1 pair (Maresh 2002)		
	Private land		1 pair (Garnett pers. comm. 2004[5])		
Palo Pinto		**1**		1 male (2002)	**1**
Brown					**unconfirmed**
Comanche					**unconfirmed**
Hood					**unconfirmed**
Hamilton		**4**			**1**
	Private land		4 males (Juarez et al. 2004)		
Johnson					**unconfirmed**
Lampasas					**1**
Mills				2 males (1989)	**2**
Parker					**unconfirmed**
Stephens					**1**
TOTAL		**2110**			**431**

REGION 2 - Edwards Plateau

County	Specific Location	Current Population[1]	Current Source by Specific Location	Texas Wildlife Diversity Database[2]	1996 Population[3]
Edwards/Kinney[4]		**265**			
	Kickapoo Caverns State Park		52 territories (Lockwood pers. comm. 2005)		
Edwards					**67**
	RM 674		135 males (Booher pers. comm. 2004[5])		
	Dobbs Run Ranch		59 males (Environmental Defense 2004[5])		
	Dobbs Mountain Ranch		17 territories (Maresh 2004a)	17 males (2003)	
Kinney				2 males (1998)	**105**
	FM 674		2 males (Maresh and Rowell 2000)		
Kerr		**436**		1 male (2001)	**602**
	Kerr Wildlife Management Area		358 males (Prochaska pers. comm. 2005)		
	Private land		78 males (Pfeffer pers. comm. 2005)		
Bandera		**28**		19 males (2000)	**48**
	Love Creek Preserve		10 males (Elliott 2004[5])		
	Hill Country State Natural Area		7 males (Lockwood and Hernandez 2000[3])		
	Private land		11 males (Wilkins et al. 2002)		
Bexar		**45**		32 males (2000)	**16**
	Camp Bullis		13 Territories (Cooksey and Thompson 2005)		
	City of San Antonio Rancho Diana		32 males (Lautzenheiser pers. comm. 2004[5])		

REGION 2 - Edwards Plateau (continued)

County	Specific Location	Current Population[1]	Current Source by Specific Location	Texas Wildlife Diversity Database[2]	1996 Population[3]
Blanco		**14**		14 males (2000)	**6**
Kimble		**35**		2 males (1996)	**26**
	Walter Buck WMA		35 males (Farquhar pers. comm. 2004[5])		
Mason		**77**		71 males (2001)	**2**
	Mason Mountain WMA		77 males (Mitchell pers. comm. 2005)		
Real		**93**		1 male (2001)	**23**
	Private land		93 males (Fushille and Ramirez 2004)		
San Saba		**11**		7 males (2001)	**22**
	Colorado Bend State Park		11 males (Lockwood and Hernandez 2001[5])		
Hays				2 males (1999)	**1**
Llano				1 male (1999)	**unconfirmed**
McCulloch		**1**		1 male (2001)	**unconfirmed**
	FM 1311		1 male (Maresh and Rowell 2000)		
Menard		**8**		8 males (2001)	**unconfirmed**
Sutton		**1**		1 male (1998)	**1**
	U.S. Highway 277		1 male (Maresh and Rowell 2000)		
Uvalde				2 males (1993)	**4**
Schleicher					**unconfirmed**
Comal					**unconfirmed**
Gillespie					**1 male seen in 1988**
Kendall					**1 male seen in 1985**
Medina		**4**		4 males (2000)	**unconfirmed**
TOTAL		**1018**			**923**

REGION 3 - Concho Valley

County	Specific Location	Current Population[1]	Current Source by Specific Location	Texas Wildlife Diversity Database[2]	1996 Population[3]
Coke		**12**		4 males (2002)	**32**
	Texas Highway 208		8 males (Maresh and Rowell 2000)		
	FM 2034		3 males (Maresh and Rowell 2000)		
	Mountain road		1 male (Maresh and Rowell 2000)		
Concho		**1**		1 male (2001)	**unconfirmed**
Nolan		**3**		1 male (1962)	**1**
	Private land		3 males (Turner pers. comm. 2004[5])		
Runnells		**2**		2 males (1998)	**5**
	County Roads 189 & 194		2 males (Maresh and Rowell 2000)		
Taylor		**125**		2 males (2003)	**1**
	Horse Hollow		60 territories (Maresh pers. comm. 2005)		
	Buffalo Gap wind farm		59 males (Maresh 2005b)		
	Camp Barkeley		6 males (Maresh 2005a)		
Tom Green		**6**		2 males (1999)	**13**
	FM 2034		4 males (Maresh and Rowell 2000)		
	Susan Peck Road		2 males (Maresh and Rowell 2000)		
Irion				3 individuals (1977)	**18**
Sterling					**1 male (last survey 1990)**
TOTAL		**149**			**70**

REGION 4 - Southwest and Trans-Pecos

County	Specific Location	Current Population[1]	Current Source by Specific Location	Texas Wildlife Diversity Database[2]	1996 Population[3]
Brewster		**15**		14 males (2003)	**16**
	Big Bend National Park		15 males (Maresh 2004c)		
	Black Gap Wildlife Management Area				
	Private land				
Terrell		**86**		26 males (2003)	**8**
	Oasis and Canon Ranches		60 pairs (Elliott pers comm. 2004[5])		
	Chandler Independence Creek Preserve		26 males (Maresh 2004b)		
Val Verde		**133**		43 males (1999)	**173**
	Devils River State Natural Area		78 males (Lockwood pers. comm. 2005)		
	Dolan Falls Ranch Preserve		55 males (Farquhar pers. comm. 2005)		
Crockett		**2**		26 males (2003)	**9**
	Texas Highway 290		1 male (Maresh pers. comm. 2004[5])		
	FM 2083/Howard Draw Road		1 male (Maresh and Rowell 2000)		
Pecos				1 individual (1975)	**3**
TOTAL		**236**			**209**

OTHER TEXAS COUNTIES

County	Specific Location	Current Population[1]	Current Source by Specific Location	Texas Wildlife Diversity Database[2]	1996 Population[3]
Callahan		**2**		2 males (2003)	**not mentioned**
Cooke					**not mentioned**
Eastland					**not mentioned**
Ellis					**not mentioned**
Grayson					**not mentioned**
McLellan					**not mentioned**
Tarrant					**not mentioned**
TOTAL		**2**			

OKLAHOMA

County	Specific Location	Current Population[1]	Current Source by Specific Location		1996 Population[3]
Comanche		**2474**			**150**
	Wichita Mountains Wildlife Refuge		2119 territories (Waldstein pers. comm. 2005)		
	Fort Sill		355 territories (Grzybowski 2005)		
Blaine		**17**			**17**
	Salt Creek Canyon area		17 males (Grzybowski 2003)		
Cleveland		**4**			**3**
	Lake Stanley Draper		4 males (Shackford 2004)		
Caddo					**last seen 1990**
Canadian					**last seen 1988**
Beaver					**not mentioned**
Creek					**not mentioned**
Dewey					**last seen 1964**

OKLAHOMA (continued)

County	Specific Location	Current Population[1]	Current Source by Specific Location		1996 Population[3]
Garvin					**last seen 1962**
Kiowa					**last seen 1963**
Logan					**last seen 1967**
Major					**not mentioned**
Murray					**not mentioned**
Oklahoma					**last seen 1984**
Payne					**not mentioned**
Tulsa					**not mentioned**
TOTAL		**2495**			**170**
U.S. TOTAL		**6,010**			**1,803**

MEXICO

State	Current Population[1]	Current Source by Specific Location
Coahuila	**139**	Farquhar et al. 2003
Nuevo Leon	**98**	Farquhar et al. 2003
Tamaulipas	**22**	Farquhar and Gonzalez 2005
MEXICO TOTAL	**259**	

GRAND TOTAL 6,269

[1] Most recent abundance estimates are those documented since 2000.

[2] TWDD as of 14 Sept. 2005. TWDD data is only included in Current Pop if year >1999 and TWDD is the only source (year of most recent observations in parentheses).

[3] Number of males documented in each county according to USFWS (1996).

[4] Fort Hood is in both Bell and Coryell Counties. Kickapoo Caverns State Park is in both Edwards and Kinney Counties.

[5] As cited in USFWS (2004).

APPENDIX B.

Museum records of black-capped vireos that were collected in Mexico outside of the breeding season.

State	Number of specimens	Museum
Sinaloa	1	Smithsonian Institution National Museum of Natural History, Washington, D.C., USA
	8	Moore Laboratory of Zoology, Occidental College, California, USA
	2	Natural History Museum of Los Angeles County, California, USA
	1	Field Museum of Natural History, Chicago, Illinois, USA
	1	University of Kansas Natural History Museum, Lawrence, Kansas, USA
	1	Delaware Museum of Natural History, Delaware, USA
	1	Museo Zoologico de la Facultad de Ciencias, Universidad Nacional de Mexico, Mexico
Durango	4	Moore Laboratory of Zoology, Occidental College, California, USA
Nayarit	3	Moore Laboratory of Zoology, Occidental College, California, USA
	7	Delaware Museum of Natural History, Delaware, USA
	2	Field Museum of Natural History, Chicago, Illinois, USA
	1	Burke Museum of Natural History, University of Washington, Seattle, Washington, USA
Jalisco	1	American Museum of Natural History, New York, USA
	2	Moore Laboratory of Zoology, Occidental College, California, USA
	1	Delaware Museum of Natural History, Delaware, USA
	2	Louisiana State University Museum of Natural History, Baton Rouge, Louisiana, USA
	1	Museo Zoologico de la Facultad de Ciencias, Universidad Nacional de Mexico, Mexico
Colima	1	Museum of Vertebrate Zoology, Univ. of California, Berkeley, California, USA
	1	Delaware Museum of Natural History, Delaware, USA
Michoacan	1	Moore Laboratory of Zoology, Occidental College, California, USA
Mexico	1	Smithsonian Institution National Museum of Natural History, Washington, D.C., USA
Guerrero	1	London Natural History Museum, United Kingdom
	1	Canadian Museum of Nature, Ottawa, Canada
Oaxaca	2	Delaware Museum of Natural History, Delaware, USA
	1	Museo Zoologico de la Facultad de Ciencias, Universidad Nacional de Mexico, Mexico