Recovery of the Golden-cheeked Warbler

FY 2007-2009

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Introduction

The activities in this final report were coordinated with the ongoing Fort Hood off-site Conservation Project (FHOSCP). Funding from Texas Parks and Wildlife directly supported data collection.

The overall goal of our work is to contribute to the environmental management goals of Fort Hood through participation in an off-site conservation program. Included are reports focused on three main sections: 1.) Distribution and Abundance of Golden-cheeked Warblers on Private Lands, 2.) Investigating Habitat Use, Habitat Selection, Reproductive Success, and Potential Factors Affecting Reproductive Success of Golden-cheeked Warblers, and 3.) Outcomes and Outreach. Activities under these projects were initiated in Fiscal Year 2005 and have continued through 2009.

Distribution, Abundance, and Breeding Activity of Golden-cheeked Warblers on Private Lands

Background

The Endangered Species research and monitoring components are designed to provide information on activities of Golden-cheeked Warblers [GCWA] and Black-capped Vireos [BCVI] on private lands. This portion supports the overall objective to quantify population status and habitat conditions for the GCWA and BCVI.

Like other songbirds, both GCWA and BCVI respond to the amount and distribution of habitat at spatial scales ranging from the broad (landscape) scale down to relatively fine (within territory) scale. Our project was designed to document habitat occupancy as well as quantify activity (e.g., breeding status), and success (e.g., number of young produced) of these species at various spatial scales, which will allow us to make specific predictions on the habitat needs of the birds and how future management practices across private lands are likely to affect their populations.

Methods

Point count surveys were conducted in Coryell and Hamilton counties beginning in 2003 and continued in 2004 and 2005 prior to the implementation of the FHOSCP. These surveys were located on private property and each survey station was systematically spaced 400 m apart on each property that was surveyed. During the 2003 – 2005 survey seasons seven focal species where observed and recorded. The seven species included GCWA, BCVI, Northern bobwhite, Brown-headed cowbird, White-eyed vireo, Bell’s vireo, and Painted bunting. Surveys were conducted from mid-March through mid-May with each survey station visited three times during the survey season. Survey protocol consisted of a 6-minute auditory survey followed by a one- minute playback of GCWA with a two-minute auditory survey then a one-minute playback of BCVI followed by another two-minute auditory survey.

At the conclusion of the 2005 survey season, all data collected for the previous three years was analyzed by Darryl MacKenzie (Proteus Wildlife Research Consultants). MacKenzie’s analysis suggested a protocol that would provide data sufficient to detect a 5% - 10% change in the occupancy of GCWA and BCVI in the study region over time. MacKenzie recommended that each survey station should be visited 6 times to accurately determine presence-absence for GCWA and BCVI. MacKenzie also suggested that the surveys should be conducted for a period of eight weeks (April 1 – May 31) and that the surveys should be completed within three hours after sunrise (Appendix A & B). It was determined that the use of the playback method and the six visits to each station increased the ability to detect warblers and vireos when they were present; it also increased certainty that when warblers and vireos were not detected, they were truly absent. Therefore, the same protocol of a six-minute auditory survey followed by playbacks for GCWA and BCVI was used as in the previous years.

We modified our survey methods for 2008 and 2009 to surveying for presence of GCWA in oak-juniper woodland patches of available habitat. We made this change in survey method to survey biologically relevant units, woodland patches rather than point count stations. We identified patches of mature oak-juniper (*Quercus* spp. - *Juniperus ashei*) woodland using classified Landsat imagery and considered these areas to be potential patches of GCWA habitat. We used a double observer removal sampling design, with 2 independent observers simultaneously surveying each patch. Once a GCWA was detected in a patch, the survey was considered complete and we did not revisit that patch. If no warblers were detected during a survey visit, we revisited the patch up to 6 times (i.e., 3 visits with 2 observers each). We also recorded any BCVI detected during surveys for GCWA.

Accomplishments

We surveyed 44 patches in 2008 and 40 patches in 2009 in Bell, Bosque, Coryell and Hamilton counties, in the Leon and Bosque River watersheds, including 22 sub-watersheds (Figure 1, Figure 2), on 34 private properties and 2 state parks, a sample region of approximately 19,700 acres. In 2007, we conducted surveys at 284 points located on 32 privately owned properties and 1 state park. GCWA occupancy was confirmed at 40.6% of the sites; and BCVI occupancy was confirmed at 4.9% of sites. Surveys in 2003 found occupancy of 13.8% for GCWA and 5.6% for BCVI at 300 sites. During 2004, we concentrated 400 survey sites in the 54,000 acre Coryell Creek drainage with resulting occupancy of 32% for GCWAs and 6.5% for BCVIs. During the 2005 survey season we surveyed 293 sites in Coryell and Hamilton counties with occupancy of 12.6% for GCWAs and 5.4% for BCVIs. During the 2006 survey season we surveyed 210 points under the new survey design and occupancy of 40.0% for GCWAs and 6.2% for BCVI were observed (Table 1). An Excel spreadsheet containing the point count information from the 2006 and 2007 survey seasons was submitted to Texas Parks and Wildlife following the survey period in 2009containing summary data with X and Y coordinates accurate to +/- 1,000 meters and metadata.

Figure 1. Map of study area indicating counties, watersheds, and sub-watersheds where we surveyed for GCWA in 2009.

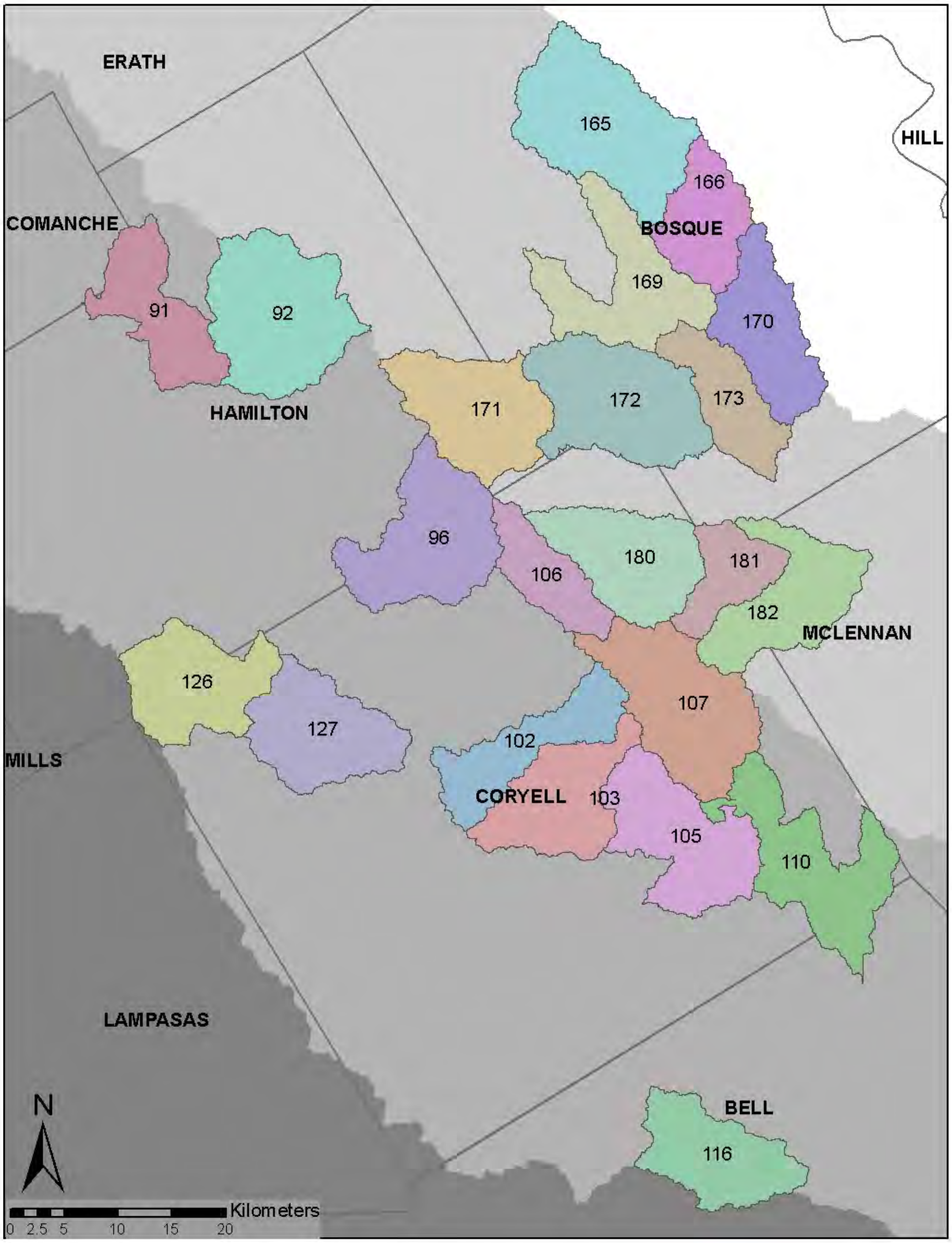


Figure 2. Legend indicating watersheds and sub-watersheds for study area map in figure 1.

Legend

Leon River Sub-watersheds

Number, name

🟦 102, Leon River Section 16 (Dodd and Stillhouse Branch)
🟥 103, Leon River Section 17 (Cottonwood, Shoal, and Turnover Creek)
🟪 105, Leon River Section 18
🟪 106, Upper Coryell Creek
🟧 107, Lower Coryell Creek
🟩 110, Leon River Section 19 (Upper Lake Belton)
🟩 116, Lower South Nolan Creek
🟨 126, Cowhouse Creek Section 4 (Dry and Langford Branch)
🟪 127, Cowhouse Creek Section 5 (Bee Creek)
🟦 165, North Bosque River Section 10 (Gibson Branch; Bosque County Reservoir)
🟪 166, North Bosque River Section 11 (Dyes and Stanifer Branch; Shumacher Creek)
🟩 169, Meridian Creek Section 1 (Bee Creek)
🟦 170, North Bosque River Section 12
🟨 171, Upper Neils Creek (Middle Fork Neils and South Fork Neils Creek)
🟩 172, Neils Creek Section 1 (Boggy and Jack Branch; Gary Creek)
🟫 173, Neils Creek Section 2 (Turkey and Shoal Creek)
🟩 180, Upper Middle Bosque River (Cave Creek)
🟪 181, Middle Bosque River Section 1 (Rainey Creek)
🟩 182, Middle Bosque River Section 2 (Bluff Creek)
🟥 91, Leon River Section 9 (Pecan Creek and Alex Branch)
🟩 92, Leon River Section 10 (Gum Branch; Indian, Little Egg, and Egg Creek)
🟪 96, Leon River Section 13 (Eagle Creek)

Watersheds

Name

⬜ **Bosque**
⬜ **Leon**
⬛ **Lampasas**

Table 1- Occupancy rates (%) for GCWA and BCVI 2003-2006 survey seasons.**

	Occupancy Rates (%)				
Year	GCWA	BCVI	# points or patches	Location (County)	# Times surveyed
2003	13.8	5.6	300	Coryell and Hamilton	3
2004	32.0	6.5	400	Coryell Creek Watershed	3
2005	12.6	5.4	293	Coryell and Hamilton	3
2006	40.0	6.2	210	Coryell and Hamilton	6
2007	40.6	4.9	284	Bell, Bosque, Coryell, and Hamilton	6
2008	77	2	44	Bell, Bosque, Coryell, and Hamilton	≤6
2009	70	8	40	Bell, Bosque, Coryell, and Hamilton	≤6

**Note: point counts conducted during the 2003, 2004 and 2005 seasons were done as part of the Leon River Restoration Project. In 2006 Texas Cooperative Extension (TCE) and Texas Parks and Wildlife Department (TPWD) signed a “Memorandum of Understanding” stating that TCE would be working as an “Agent” of TPWD for work associated with the Fort Hood Off-site Conservation Project. Therefore, information collected by TCE is covered under TPWD code and will remain confidential.

Investigating Habitat Use, Habitat Selection, Reproductive Success, and Potential Factors Affecting Reproductive Success of Golden-cheeked Warblers

Summaries of previous research conducted prior to 2009 can be found in the appendices. Jerrod Butcher investigated patch size thresholds of reproductive success of GCWA in 2005–2007 (Appendix C & D). Shannon Farrell investigated on the relationships between frequency of cowbird parasitism on BCVI and white-eyed vireos and characteristics of the neighboring avian assemblage in 2005–2006 (Appendix E) Andrew Campomizzi investigated the impact of red imported fire ants on songbird nest survival in 2006–2007 (Appendix F & G).

Brown-headed Cowbird Trapping

Brown-headed cowbirds (*Molothrus ater* [BHCO]) are brood parasites that impact many avian host species (Figure 3). Based on the research conducted in 2006 on BHCO parasitism in BCVI, we developed a study designed to determine the effects of trapping and removing BHCO during the 2007-2009 field seasons.

Methods

We ran 30 cowbird traps on 7 properties located in Coryell County in 2007. We monitored 12 BCVI nests and 26 white-eyed vireo nests in areas with cowbird. We used 3 additional properties with no cowbird trapping being conducted for control where we monitored 14 BCVI nests nests and 9 white-eyed vireo nests.

We ran 33 cowbird traps on 8 properties located in Coryell County in 2009. We monitored 5 BCVI nests and 25 white-eyed vireo nests in areas with cowbird trapping. We used 5 additional properties with no cowbird trapping being conducted for control where we monitored 5 BCVI nests and 36 white-eyed vireo nests.

We ran 33 cowbird traps on 8 properties located in Coryell County in 2007. We monitored 12 BCVI nests and 17 white-eyed vireo nests in areas with cowbird trapping. We used 7 additional properties with no cowbird trapping being conducted for control where we monitored 18 BCVI nests and 49 white-eyed vireo nests.

Results

We trapped and removed BHCO from 22 March – 8 June 2007, 1 March – 27 June 2008, and 2 March – 30 June 2009. We removed 1,650 BHCO from the 30 traps on 7 properties in Coryell county in 2007, 1,846 BHCO from the 33 traps on 8 properties in Coryell county in 2008, and 1,136 BHCO from the 33 traps on 8 properties in Coryell county in 2009 (Table 2).

Areas with cowbird trapping had higher success for both BCVI and white-eyed vireos. In 2007, parasitism frequency was 33% on BCVI and 12% on white-eyed vireos on properties with cowbird trapping and 93% and 44% respectively on properties without cowbird trapping (Table 3). In 2008, parasitism frequency was 0% on BCVI and 20% on white-eyed vireos on properties with cowbird trapping and 60% and 28% respectively on properties without cowbird trapping (Table 4). In 2009 parasitism frequency was 42% on BCVI and 24% on white-eyed vireos on properties with cowbird trapping and 50% and 45% respectively on properties without cowbird trapping (Table 5).

Table 2. Number of female and male BHCO removed from traps each 2007–2009.

Year	Female	Male	Total
2007	755	895	1650
2008	731	1115	1846
2009	358	778	1136

Table 3. Frequency of BHCO parasitism on BCVI and white-eyed vireo nests in areas with and without cowbird trapping in 2007**.**

	Black-capped vireo		**White-eyed vireo**	
	Parasitized	Not Parasitized	Parasitized	Not Parasitized
Trapped	33% (*n*=12)	67% (*n*=12)	12% (*n*=26)	88% (*n*=26)
Control	93% (*n*=14)	7% (*n*=14)	44% (*n*=9)	46% (*n*=9)

Table 4. Frequency of BHCO parasitism on BCVI and white-eyed vireo nests in areas with and without cowbird trapping in 2008.

	Black-capped vireo		**White-eyed vireo**	
	Parasitized	Not Parasitized	Parasitized	Not Parasitized
Trapped	0% (*n*=5)	100% (*n*=5)	20% (*n*=25)	80% (*n*=25)
Control	60% (*n*=5)	40% (*n*=5)	28% (*n*=36)	72% (*n*=36)

Table 5. Frequency of BHCO parasitism on BCVI and white-eyed vireo nests in areas with and without cowbird trapping in 2009.

	Black-capped vireo		**White-eyed vireo**	
	Parasitized	Not Parasitized	Parasitized	Not Parasitized
Trapped	42% (*n*=12)	58% (*n*=12)	24% (*n*=17)	76% (*n*=17)
Control	50% (*n*=18)	50% (*n*=18)	45% (*n*=49)	55% (*n*=49)

Figure 3. Black-capped vireo nest with 3 brown-headed cowbird eggs.

Nest predation of black-capped vireos and neighboring songbirds in Texas

Predation has been cited as the leading cause of nest failure in songbirds (Martin 1993, Grzybowski 1995, Schmidt and Whelan 1999). However, nest failure may vary across a habitat area and a landscape, and may vary among nests due to risk factors including nest site characteristics, parental behavior, and predator behavior. Understanding of the relationship between nest success and predation is critical for developing effective conservation plans, particularly for threatened and endangered avian species, such as the BCVI, who may respond differently to nest predation than common generalist species, including the congeneric white-eyed vireo (*Vireo griseus*, [WEVI]), but predator research is still limited.

Methods

We located nests using behavioral observations and systematic searches and monitored active nests at all locations using standard visual checks every 2-7 days. We deployed a video camera system with infrared lighting and a digital video recorder

(DVR) at sample nests to identify nest predators and nest fate. We also collected vegetation and concealment data at each nest.

Results

We monitored 43 BCVI nests and 54 WEVI nests (Table 6). Only 24% of all BCVI nests (and 29% of camera-monitored nests) fledged at least one host offspring. WEVI nests were more successful, with 46% total nests fledging at least one offspring. We recorded 23 predation events by >6 predator species (Table 7). Brown-headed cowbirds and snake species were the most frequent nest predators recorded, accounting for 74% of all predation events. Additionally, BHCO only depredated non-parasitized nests. For camera nests, there was no significant relationship between nest fate (fledge vs. fail) and concealment at the nest (BCVI $p = 0.590$; WEVI $p = 0.590$) or distance to edge (BCVI $p = 0.773$; WEVI $p = 0.751$). However, there may be a relationship between nest predator species and distance to edge or percentage of vegetation concealment. The results show that identified nest predator species differed from previous research on Ft. Hood where snakes and red imported fire ants (*Solenopsis invicta*) were cited as the major predators. Importance of vegetation concealment may also factor into the different search strategies implemented by predator species.

Additionally, the number of predation events by BHCO demonstrates the potential for this species to have multi-level impacts on vireo productivity, even with active cowbird management. Some properties have ongoing BHCO trapping programs which decreased parasitism, but we still observed high levels of BHCO nest predation on some of these sites. Our results will further our understanding of nest predator assemblages on public and private land, leading to increased effectiveness of future recovery efforts for BCVI and other species in the region including GCWA.

Table 6. Nest fates of monitored black-capped vireo and white-eyed vireo nests in 2008 and 2009**.**

	BCVI-All nests				BCVI-Camera nests				WEVI-camera nests			
	2008		2009		2008		2009		2008		2009	
	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*
Abandoned	40.0	4	36.4	12	37.5	3	39.1	9	10.0	3	8.7	2
Depredated	40.0	4	39.3	13	37.5	3	26.1	6	43.3	13	26.1	6
Fledged	20.0	2	21.2	7	25.0	2	30.4	7	46.7	14	65.2	15
Unknown	0.0	0	3.0	1	0.0	0	4.3	1	3.2	1	0.0	0
Parasitized	20.0	2	45.4	15	25.0	2	43.4	10	30.0	9	21.7	5

Table 7. Identified nest predator species at BCVI and WEVI nests in Coryell Co in 2008 and 2009.

		BCVI			WEVI		
Predator		2008	2009	Total	2008	2009	Total
Brown-headed Cowbird	*Molothrus ater*	2	1	3	4	2	6
Snake spp.	--	1	2	3	3	2	5
Ant spp.	*Solenopsis invicta*	0	1	1	1	1	2
Western Scrub-Jay	*Aphelocoma californica*	0	1	1	0	0	0
Hawk spp	*Accipiter spp.*	0	0	0	1	0	1
Raccoon	*Procyon lotor*	0	0	0	0	1	1
Totals		3	5	8	9	6	15
Unknown		1	0	1	0	1	1
Predation not recorded		0	2	2	6	1	7

Influence of Extra-pair Paternity and Use of Public Information for Breeding Site Fidelity in Songbirds

Extra-pair paternity may have an important role in the use of public information (i.e., observable cues from other individuals that may indicate habitat quality) for breeding site fidelity in songbirds. We are investigating whether songbirds use public information for site fidelity decisions even when that information does not indicate reproductive success through extra-pair paternity. We are testing (1) whether adult males use public information cues for breeding site fidelity when fledglings in adjacent territories are not their genetic offspring, (2) whether adult males use public information more often than adult females, and (3) relative importance of public information, personal reproductive success with social mate, and extra-pair paternity in explaining patterns of site fidelity for males and females. Our results will improve our understanding of why songbirds breed in particular locations and how individuals use personal and public information of productivity for site fidelity decisions. Improving our understanding of site fidelity will inform land managers about what kinds of reproductive success are needed to maintain breeding areas for migratory songbirds.

Methods

We are investigating site fidelity patterns of white-eyed vireos (*Vireo griseus*) by territory mapping, nest searching, color-banding, and genotyping individuals breeding in an 80 ha patch of woodland in central Texas from 2008 to 2010.

Results

We monitored white-eyed vireos in 27 and 40 territories in 2008 and 2009, respectively. We collected DNA from 67 adults and 101 nestlings for genotyping in 2010. We ran preliminary paternity analyses at in the genetics lab at Purdue University during fall 2008 to ensure field sampling techniques were appropriate. This project has 1 field season remaining and will be completed in spring 2011.

Use of social information for habitat selection in Golden-cheeked warblers

Most habitat selection research has focused on association between species occurrence and vegetative and geologic characteristics; where behavior is considered, the focus has been on competitive interactions, density dependence, and exploitative resource competition. However, positive intraspecific interactions, such as attraction to inadvertent social information in the form of auditory or visual location cues from conspecifics, may be an important part of habitat selection for many taxa including migratory songbirds. We conducted a replicated, manipulative field experiment on 16 pairs of sample units in oak-juniper woodland patches with a range of canopy cover in Texas in 2008 and 2009, to investigate the use of conspecific cues on territory selection by GCWA, and the consequent reproductive success in the selected sites. Identifying information used for habitat selection decisions is critical for understanding resulting patterns of habitat use, identifying fitness consequences of decisions, predicting impacts of anthropogenic changes to habitat, and creating accurate predictive habitat models for management of migratory songbirds including GCWA.

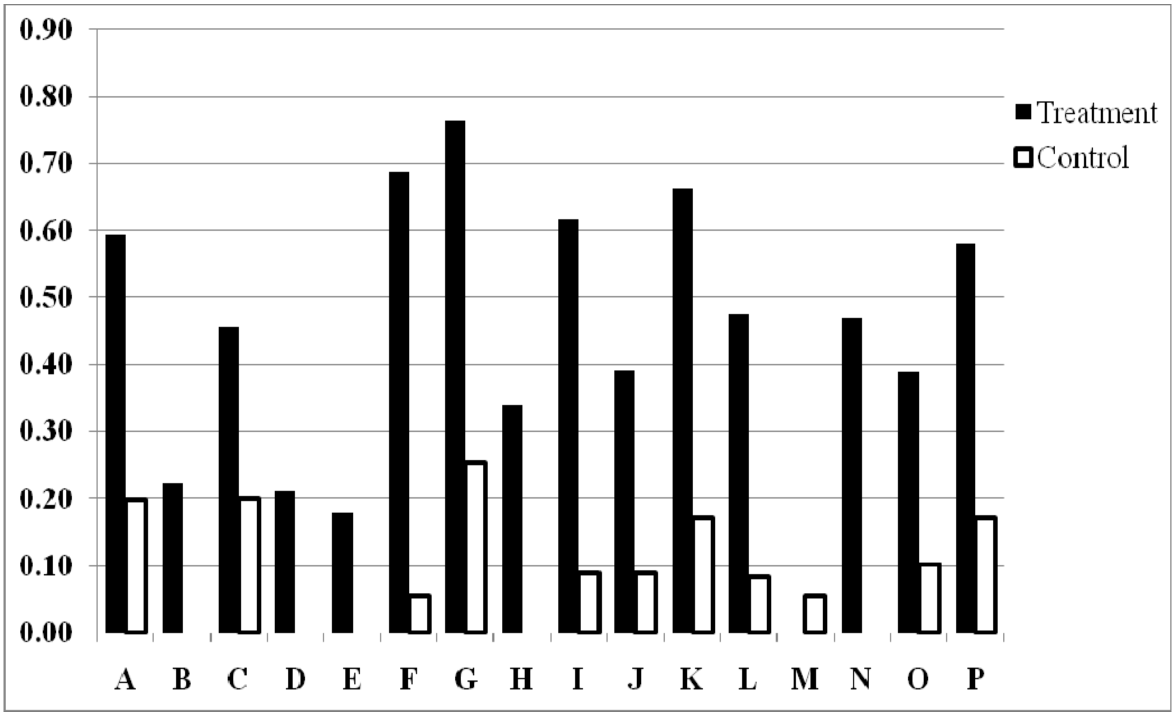
Methods

We identified potential pairs of sample units of oak-juniper woodland patches with canopy cover ranging from ≈30%–95% cover in Coryell and Bosque Counties. Sample units were comprised of a 250-m-radius circle around a central point. We randomly assigned treatment to one sample unit in each pair. Treatment consisted of broadcast stations that played a loop of songs, calls, and periods of quiet from ≈ 06:00–10:00 every day for 2 weeks prior to expected bird arrival and for 4 weeks following arrival of the first male GCWA. Control sites did not have broadcast stations. We conducted territory mapping in each sample patch ≥1 time per week from 15 March – 15 May, by recording GPS locations of singing males, females, or pairs observed at each territory visit across at least 3 visits to the territory. We conducted fledgling search surveys 1 time per week for 1-hr in each territory 1 May–15 July to determine productivity. We generated minimum convex polygons to delineate territories. We calculated territory density as number of territories within each 250-m radius buffer divided by the total woodland patch area within the 250-m radius buffer, excluding non-woodland areas (e.g., adjacent pasture).

Results

Territory density was 5 times higher in treatment units ($p = 0.001$) (Figure 4); sign tests show the probability that 15 of 16 treated plots had higher density than controls was 0.02, suggesting the results differ significantly from chance. Territory density increased in response to treatment even in low canopy cover woodland typically considered poor or suboptimal habitat. We are currently analyzing reproductive success and fledging data in relation to treatment and woodland canopy cover. In 2010, we will replicate this study on other study areas. Additionally, in 2009, we began conducting and experiment to investigate the use of conspecific vocal cues present during the post-breeding period in habitat selection decisions made by warbles in the subsequent year. We simulated vocalizations during the post-breeding period of 2009 and will collect response data for this, following the same methods described above, in March–May 2010.

Figure 4. Territory density of GCWA in treatment and control paired sample units in 2008 (sample pairs A to E) and 2009 (sample pairs F to P).



Outcomes and Outreach

The outcome and results of the research and studies in this project have provided information and opportunities to support additional outreach efforts. These efforts seek to increase the overall knowledge and application of the science to address real world problems.

Golden-cheeked Warbler Habitat Modeling

Previous research utilized measures of vegetation, geology, and geomorphology to map predicted habitat for GCWA. Based on this background work and survey data collected since 2003 an updated habitat model was created. Analysis of previous data showed that the amount of oak-juniper (*Quercus-Juniperus*) woodland in the landscape was the only characteristic that was correlated with GCWA presence. A LANDSAT layer classified by the University of Texas Center for Space Research (UTCSR) was used to identify areas of oak-juniper composition and map locations of oak-juniper woodland across the landscape in Hamilton, Lampasas, Bosque, Coryell, and Bell counties. Point count information from the 2006 survey season was used to evaluate the accuracy of the predictive model. Upon analysis it was determined that 82% of survey stations in >60% woodland composition were occupied by GCWA, 64% of stations in the 40-60% woodland composition were occupied, and 20% of the stations in <40% woodland composition were occupied. (Appendix H)

Monitoring Golden-cheeked Warblers on Private Lands in Texas

Because the majority of GCWA habitat exists on private lands, monitoring strategies must focus on habitat in these private holdings. By outlining study designs and protocols using repeated presence–absence surveys across a gradient of patch sizes, we developed a range-wide monitoring program for the warbler in Texas. We surveyed 200–400 point-count locations across approximately 30 private properties annually from 2005 to 2008. We estimated patch dynamics and associated detection probabilities for GCWA. Patch size had a strong association with patch occupancy, and all patches >160 ha were predicted to be occupied. We found no evidence that large GCWA populations located on public lands in the vicinity of our study area influenced occupancy dynamics. We performed simulations that indicated that removal-based sampling is superior to standard sampling. Based on our results, surveying GCWA presence in oak–juniper patches under a removal modeling framework should be considered as one alternative for range-wide monitoring programs because patch-level monitoring would be necessary to estimate proportion of range occupied. Large contiguous patches are rare across the species' range; hence, conservation and management of the mosaic of smaller patches within a landscape context would be required for maintaining species viability (Appendix I).

Statewide Surveys for Golden-cheeked Warblers

The purpose of the Statewide Project is to determine the distribution and abundance of GCWA throughout its breeding range in central Texas through the development of a GCWA occupancy model. Current population estimates for GCWA are highly variable and based on few sampling points relative to the extent and variability in potential breeding habitat. In the spring of 2008 and 2009, we surveyed hundreds of points throughout the species' breeding range on both public and private properties in an effort to more accurately estimate the current distributions and abundances of the birds. Increasing the accuracy of population distribution and abundance estimates will allow for a better understanding of the trends in rangewide populations over time. In addition, we are examining patch-level occupancy estimates relative to certain habitat characteristics. The surveys and analyses will ultimately result in refined habitat suitability maps for GCWA breeding habitat in Texas, increased reliability of habitat availability predictions, and recommendations of sampling protocols for long-term monitoring.

Methods

Given the endemism of GCWA to patches of oak-juniper woodlands, we defined oak-juniper patches as being the operational sampling unit for determining occupancy and abundance. We delineated oak-juniper patches using supervised classifications of Landsat Thematic Mapper images with ERDAS IMAGINE 9.2 (ERDAS Inc., Norcross, Georgia, USA) for the entire 35-county range of the GCWA in central Texas. Of the resulting oak-juniper patches, we randomly selected ~250 patches proportional to patch size classes, percent landscape composition classes, and amount of potential habitat encompassed by each GCWA recovery region. Approximately 95% of the randomly selected patches were located on private property.

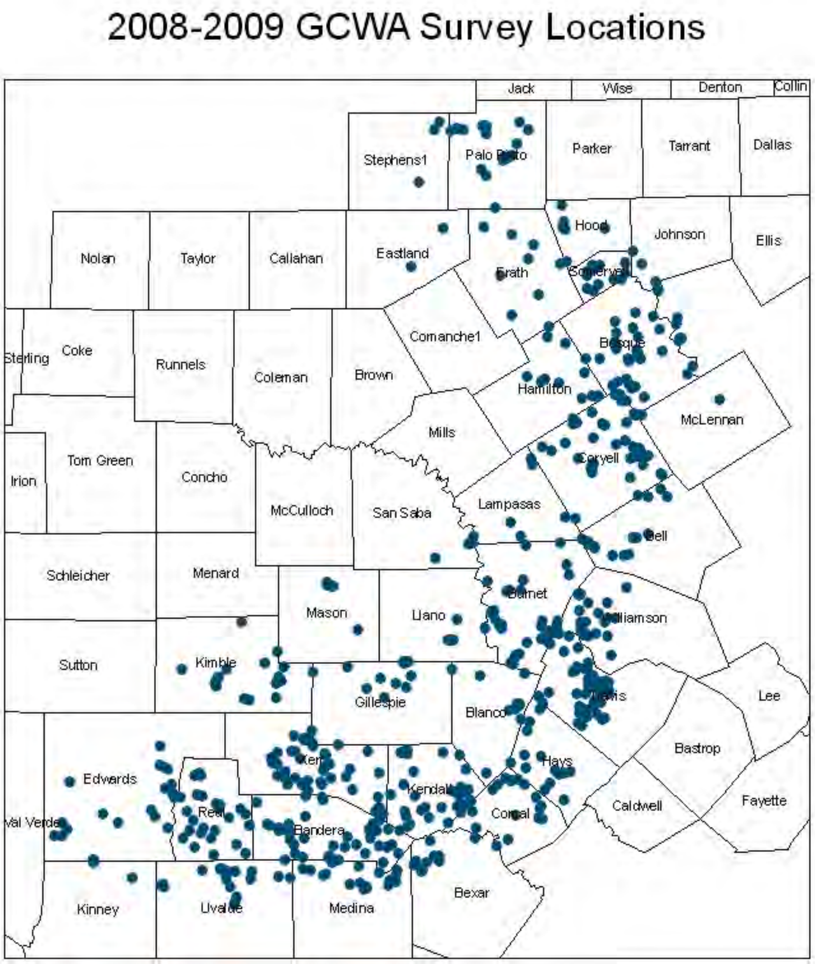
We attempted to conduct occupancy surveys on as many of the randomly selected habitat patches as possible from mid-March through the end of May 2009. We conducted occupancy surveys using a removal sampling design with 2 independent observers simultaneously surveying each patch. We did not broadcast recorded GCWA calls during the occupancy survey. Once a GCWA was detected within a patch (e.g., presence was established), that patch was not revisited for occupancy surveys. However, if no GCWA were detected during a visit, the patch was revisited up to 6 total visits (i.e., 3 visits each by 2 observers) to estimate presence, absence, and GCWA detectability.

For abundance surveys, we used ArcMap 9.2 (ESRI, Redlands, California, USA) to assign point count stations randomly within each oak-juniper patch; each point was 400-600 m from all other points and ≥30 m from the patch or region edge. The number of points per patch varied proportional to patch size. We conducted double observer, fixed-radius point count surveys at each point (i.e., point count station) in the patch provided a GCWA had been detected during the occupancy surveys. Two independent observers surveyed each point count station together, randomly determining who had the role of Observer 1 and Observer 2 at each point count station. Observer 1 communicated to Observer 2 all birds seen and heard and their distance from the point (0–50 m or >50–100 m). Observer 2 recorded all birds detected by Observer 1, along with surveying the area and recording birds not detected by Observer 1. Point count stations were visited 1 time by the 2 observers between mid-March and late May 2009. Observers did not broadcast recorded warbler calls during the abundance survey. The actual locations of warblers were not recorded during the survey; rather, the data must be understood as a warbler occurring within 100 m of the point count station.

Results

We surveyed 603 patches in 2008 and 2009 (Figure 5). A golden-cheeked warbler was detected at 375 (62.2 %) of these patches during occupancy surveys. Patches where occupancy was determined were then sampled by point counts. Of 1105 total points counted within these patches, a golden-cheeked warbler was detected at 525 points (47.5 %). Approximately 90% (540) of the patches we surveyed were located on private lands.

Figure 5. Locations of patches surveyed for GCWA during 2008 and 2009.



Case Study on Conspecific Attraction

Wildlife biologist use knowledge about wildlife-habitat relationships to create habitat models to predict species occurrence across a landscape. This study looked at the spatial relationships and conspecific attraction influences. Analysis of data suggested that conspecific attraction can lead to clustered distribution of wildlife within available habitat, reducing the predictive ability of habitat models based on vegetative and geographic parameters alone. (Appendix J)

Appendix A

Analysis of existing occupancy data for black-capped vireo and golden-cheeked warbler

Analysis of Existing Occupancy Data for
Black-capped Vireo and
Golden-cheeked Warbler

Darryl I. MacKenzie

7 March 2006

Proteus
wildlife research consultants

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Executive Summary

- Survey at least 300 stations 6 times per year
- Trial alternative survey protocols this year in an effort to increase detection probability. The order in which the protocols are used at each station should be randomized.
- Detection probability for golden-cheeked warblers appears to vary by time of day (higher between 0700 and 0930, peaking at 0800). Suggest that each station be surveyed at different times of the day during the year to avoid introducing a form of heterogeneity.

Introduction

Surveys for golden-cheeked warblers (GCWA) and black-capped vireos (BCVI) were conducted near Fort Hood from 2003-2005. The protocol for these surveys comprised of a six minute passive listening period followed by a playback and 2 minute listening period for each of the target species not detected in the initial listening period. However recent e-mail discussions have highlighted that this protocol may not always have been followed, and that information on whether a playback was used, is not readily available (A. Campomizzi, *personal communication*). Monitoring stations were visited up to 3 times per year, but the same stations were not necessarily visited in consecutive years.

The intent of this report is to analyze this previously collected data to provide advice on the sampling protocol that should be used for this upcoming season. A main focus here was to investigate factors that may affect the ability of surveyors to detect the species in the field (i.e., factors that affect detection probability).

Methods

The multi-season (year) occupancy model developed by MacKenzie et al. (2003) was used here to analyze the available data. This model allows for changes in the presence/absence of the species between years, and also for the imperfect detection of the species at a station where the species is present within a year. Because only relatively few stations were monitored in consecutive years, there will be limited information with respect the processes of colonization and local extinctions between years hence parameterizations of the model were used where it was assumed that occupancy changes at random between years or no changes occurred during the 3 years (MacKenzie et al. 2005).

Model selection with respect to the occurrence of each species was limited to general comparisons of whether occupancy appeared to be; 1) different each year (with random changes); 2) constant for the 3 years (but with any changes occurring at random); and 3) no changes in occupancy between years. However, as the spatial coverage of the monitoring varied substantially between years, this aspect of the results should be interpreted with some caution. No model selection was performed on the factors that may influence occupancy to maintain a relatively general model in this respect while assessing the importance of factors that may influence detection probability. The factors that were included in the models that may affect the probability of occurrence of the species were; 1) whether the station was located within the Coryell watershed (CW; as the spatial extent of the surveying in 2004 was essentially limited to the Coryell watershed); 2) percent total canopy cover (%C); and 3) percent slope (%S). The factors %C and %S were standardized such that they had a mean of 0 and standard deviation of 1.

Following an assessment of how the occurrence of the species may generally change over time, factors that may affect the detection of the species were investigated. The factors that were considered as candidates to influence detection probability were; 1) year (Y); 2) time of day (modeled as a 3rd-order polynomial; TOD); 3) CW; 4) %C; and 5) %S. Because of the potentially large number of possible models, and the difficulties in defining a subset of biologically plausible models (or more correctly, the difficulties in defining which models might not be biologically plausible), a simple “leave one out” approach to model selection was used. Each factor was removed from the most general model and if the AIC value for the resulting model was (approximately) 2 points smaller than the AIC value for the general model, that factor was deemed relatively unimportant for modeling detection probability. Following this process of removing each factor, the final model was obtained by including only the “important” variables. While somewhat ad-hoc, given the exploratory nature of this analysis, this approach seems reasonable.

Using the results from the analysis of the GCWA and BCVI data, a small simulation study was used to compare possible study designs for monitoring a trend in occupancy over a 5-year period. For simplicity, the parameterization of the multi-season model that assumes changes in the occupancy state of a station between years are random was used. Based upon the results of the initial analysis, a number of scenarios were investigated. For each scenario, 5000 sets of data were simulated from which the standard error of the trend parameter was approximated. Using the approximated standard error, a two-sided z-test was used to approximate the power of a likelihood ratio test for trend (with $\alpha = 5\%$ or 10%). In all scenarios it was assumed data was collected for a 5-year monitoring period.

Results

Golden-cheeked Warbler

Fitting the 3 general models for occupancy, the model which assumed the occupancy state of sites was static for the 3 years had a very high level of support compared to models that allowed occupancy to change at random (ΔAIC > 20). The static occupancy model suggests that at a station with average total canopy cover (79%) and slope (8%), the probability of occupancy is 0.30 (SE=0.07) outside of the Coryell Watershed and 0.50 (0.05) inside the Coryell watershed. There is not strong evidence that occupancy varies with level of canopy cover or slope as the estimated SE's are relatively large compared to the estimated magnitude of the effect, although there is an indication of a positive relationship between occupancy and both of these factors. The estimated effect on the *odds* of occupancy (see MacKenzie et al. 2005: p 71-75 for a definition) is that a 23% (i.e., 1 standard deviation) increase in total canopy cover will multiply the odds by 1.20 (95% CI = 0.85 – 1.71), and that a 5% increase in slope will multiply the odds of occupancy by 1.24 (0.94 – 1.64).

Table 1 summarizes the results of removing each factor from the detection probability component of the model in terms of the relative change in AIC values. Only the removal of the factor CW leads to a negligible reduction in AIC, therefore detection probability was models as a function of all the factors considered. Figure 1 illustrates how the predicted detection probability changes during the day for each of the 3 years, inside and outside of the Coryell watershed, however, the estimated odds ratio associated with CW has a relatively large degree of uncertainty (estimate = 1.51; 95% CI = 0.75 – 3.08). The effect on the *odds* of detecting GCWA in a survey is that a 29% increase in total canopy cover multiplies the odds by 1.77 (1.37 – 2.29) and a 5% increase in slope multiplies the odds by 1.30 (1.08 – 1.56). A histogram of the predicted detection probabilities for the surveys conducted from 2003-2005 (as outputted by PRESENCE) is presented in Figure 2, which suggests the average detection probability for the surveys conducted from 2003-2004 was about 0.2. To assess the effect of pre/post playback, a similar model (although with the effect of CW removed for simplicity) was fit to the data where each survey was split into whether detection occurred pre or post playback. Similar inferences result, with the main result being that the odds of detecting GCWA post playback was only 0.36 (0.27 - 0.48) of the odds pre playback. This effect is also illustrated in Figure 3.

Black-capped Vireo

Fitting the 3 general models for occupancy, the model which assumed the overall probability of occupancy was constant in all 3 years, but with random changes between years, had a high level of support compared to the other models (AIC > 5). The model suggests that at a station with average total canopy cover (79%) and slope (8%), the probability of occupancy is 0.15 (SE=0.10) outside of the Coryell Watershed and 0.12 (0.04) inside the Coryell watershed. Occupancy was positively related with percent total canopy cover and percent slope. The estimated effect on the odds of occupancy is that a 23% (i.e., 1 standard deviation) increase in total canopy cover will multiply the odds by 2.33 (1.27 – 4.27), and that a 5% increase in slope will multiply the odds of occupancy by 1.50 (1.08 – 2.08).

Table 2 summarizes the results of removing each factor from the detection probability component of the model in terms of the relative change in AIC values. The removal of the factors time for day and percent slope lead to relative large reductions in AIC, with the removal of other factors resulting in negligible changes in AIC. Therefore, a model where detection probability varied by year, CW and percent total canopy cover was decided upon. The estimated detection probabilities at a site with an average percent total canopy cover, for each year, inside and outside of the Coryell watershed is given in Table 3. The effect on the odds of detecting BCVI in a survey is that a 29% increase in total canopy cover multiplies the odds by 0.62 (0.37 – 1.03). A histogram of the predicted detection probabilities for the surveys conducted from 2003-2005 (as outputted by PRESENCE) is presented in Figure 4, which suggests the average detection probability for the surveys conducted from 2003-2004 was about 0.15. Fitting a similar model to the data where each survey has been split into pre/post playback provides similar inferences, with the main result being that the odds of detecting BCVI post playback was 1.36 (0.80 – 2.31) of the odds pre playback. This effect is also illustrated in Table 4.

Simulation Study

From the above results a number of scenarios were considered to assess the ability of a 5-year study to identify a trend in occupancy. In each, detection probability was assumed to be 0.2 with either 5, 6 or 7 surveys per year. The initial level of occupancy was assumed to be 0.15 or 0.3 with either no, increasing/decreasing moderate or increasing/decreasing strong trend (i.e., trend parameters of 0, +/- 0.1 or +/- 0.2 on the logistic scale; see Figure 5). The number of stations was set at 100, 200, 400 or 600.

Figures 6-11 present the results of the simulation study. As would be expected, by increasing the number of stations surveyed each year, the trend standard error decreases and power to detect a trend increases. Interestingly, when the initial probability of occupancy is low (0.15) the standard error varies with the magnitude of the “true” trend in occupancy (Figures 6-8), but is much less variable with a higher initial probability of occupancy (0.3; Figures 9-11). In all cases, increasing the number of surveys within a year reduces the standard error (and hence increases power), but at the expense of increasing the total number of surveys conducted. However by comparing scenarios with approximately the same total level of effort (e.g., surveying 600 stations 5 times per year vs. 400 stations 7 times), note that surveying fewer stations more intensively leads to a (slightly) more precise estimate of trend.

Discussion and Recommendations

One of the main findings of the above analysis is that both GCWA and BCVI have generally low detection probabilities. In terms of designing a robust monitoring program, the low detection probability in combination with the low level of occupancy suggests that 3 surveys per year is likely to be insufficient. Based upon the work of MacKenzie and Royle (2005), the above results suggest that a more efficient monitoring program may be achieved by increasing the number of surveys per year to approximately 6, even to the extent of reducing the number of stations surveyed. This result was the basis for using 5-7 surveys per year in the simulation study.

An alternative to increasing the number of surveys per year is to increase the probability of detection per survey. Some possible methods of increasing detection probability would be to use multiple observers or increase the length of the surveys.

An interesting result for GCWA was the lower detection probability post-playback. A natural interpretation here is that the playback actually reduces detection probability for GCWA, however an alternative interpretation is that the detectability may simply decrease the longer an observer stays at a station; possibly due to disturbance or that GCWA are more vocal when an observer first enters a station (i.e., increased vocalization due to a perceived intruder to their territory). This may also be an artificial result as, as noted above, the data supplied here did not differentiate between occasions when playback surveys were/were not used (the required information was not readily available). Hence in some instances, the recorded “non-detection” of the species during the playback survey was actually due to the fact that the playback survey was never used. This will result in the post-playback detection probability being underestimated, and highlights the importance of accurately recording all relevant information.

For GCWA, the results suggest that detection probability is higher between the hours of 0700 and 0930, peaking at approximately 0800. One aspect of this result is that if stations tend to be surveyed at the same time of day, a form of heterogeneity in detection probability may be introduced. That is, stations that always tend to be surveyed at 0800 will have a higher detection probability than stations that always tend to be surveyed at 0700. This heterogeneity may introduce some bias to resulting estimates if not accounted for. Modeling this aspect of detection probability may mitigate some of this effect, although time of day and station-specific effects may be partially confounded. This potential confounding may be simply removed by ensuring that a station is surveyed at different times of day during the repeated surveys. One way of achieving this (for example) would be to randomize the order in which stations are surveyed each day.

I am mindful of the fact that 6 surveys per year of each station constitutes a relatively large increase in effort compared to previous years. I therefore suggest that this year some effort be put into comparing different survey protocols in an effort to improve the probability of detection within a survey. Following recent discussions with Dr. Michael Morrison and Mr Andrew Campomizzi, one suggestion is to compare 3 different survey protocols; 1) the status quo; 2) survey for the same total period as present, but do not use the playback; and 3) a 20 minute survey with no playback. Each protocol should be used twice per station this season, with the order randomized for each station. While there may be the potential for some confounding between survey protocol and time of day if both factors are randomized independently (e.g., first randomize the order in which a set of stations will be surveyed on each of the 6 survey days, then for each station randomize the order in which the 3 survey protocols will be used), the likelihood that any confounding would notably influence the outcome of the monitoring is probably very small due to the number of stations that will need to be monitored (i.e., the possibility that all surveys of a particular protocol are always conducted early in the day will be very remote), especially if any potential effect of these factors is modeled as part of the estimation procedure. However, if considered necessary, some form of blocking could be used to obtain a balanced design although further specifics of the logistics of the monitoring will be required (e.g., number of sites to be monitored each day).

The results of the simulation study suggest that the power to detect moderate trends in occupancy will be relatively low, even with up to 600 stations being monitored per year for 5 years. Increasing the acceptable probability of a type I error (α) improves power, but at the cost of increasing the chances of false declaring a trend in occupancy. The power to detect stronger trends is reasonable (>60%) with 200 stations. The relative costs and benefits of trading off power and type I error rate should be considered. It is also worth noting that the power to detect a trend increased as the number of surveys per site increased. This is undoubtedly a result of the decreased ambiguity as to whether a site may be occupied within a year by surveying a site more intensively. One aspect that has not been systematically considered here is increasing the time period over which the monitoring is conducted. The number of stations could be reduced by collecting data for a longer timeframe (MacKenzie 2005), although this increased timeframe may hamper ones ability to efficiently manage the population. Given the level of historic field effort that has been available, it is recommended that at least 300 stations be monitored annually for 5 years (with 6 surveys per year).

Due to the exploratory nature of this analysis, potential information on the factors that affect occupancy has not been utilized in terms of improvements to the monitoring program. The results of MacKenzie and Royle (2005) indicate that fewer surveys per year are required at stations with lower occupancy probability. Therefore, the monitoring program could be tailored such that fewer surveys are conducted at stations with low predicted occupancy, and more at stations with higher predicted occupancy. I recommend that this aspect of the monitoring program be considered once better information on the factors that affect detection has been collected.

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MacKenzie, D.I., and J.A. Royle. 2005. Designing occupancy studies: general advice and allocating survey effort. Journal of Applied Ecology 42: 1105-1114

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MacKenzie, D.I., J.D. Nichols, J.A. Royle, K.H. Pollock, L.L. Bailey and J.E. Hines. 2005. Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. Elsevier, San Diego, CA.

Table 1: Effect of removing factors from the detection probability component of the model on the relative difference in AIC values compared to the full model for GCWA.

Factor Removed	ΔAIC
Year	16.32
Time of Day	3.12
CW	-0.57
%C	18.33
%S	6.09

Table 2: Effect of removing factors from the detection probability component of the model on the relative difference in AIC values compared to the full model for BCVI.

Factor Removed	ΔAIC
Year	-0.69
Time of Day	-5.46
CW	0.10
%C	-0.53
%S	-1.94

Table 3: Estimated detection probabilities (and standard errors) for BCVI inside and outside the Coryell watershed (CW).

	Outside CW	Inside CW
2003	0.11 (0.07)	0.28 (0.11)
2004	0.04 (0.03)	0.13 (0.04)
2005	0.08 (0.06)	0.22 (0.08)

Table 4: Estimated detection probabilities (and standard errors) for BCVI inside and outside the Coryell watershed (CW), pre and post playback.

	Outside CW		Inside CW	
	Pre	Post	Pre	Post
2003	0.04 (0.03)	0.05 (0.03)	0.11 (0.07)	0.14 (0.06)
2004	0.02 (0.01)	0.02 (0.02)	0.05 (0.04)	0.07 (0.03)
2005	0.03 (0.02)	0.04 (0.03)	0.08 (0.03)	0.11 (0.04)

Figure 1: Illustrating the effects of time of day and whether station was located within the Coryell watershed on detection probability for GCWA.

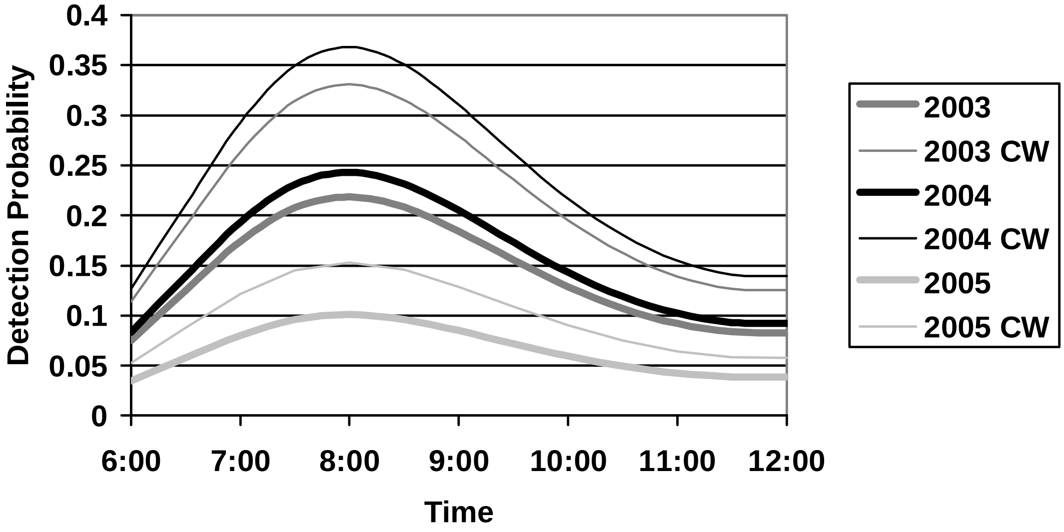


Figure 2: Histogram of the estimated detection probabilities for all surveys conducted from 2003-2005 for GCWA from PRESENCE.

```
Distribution of p's:
  0.00    27:****
  0.03   192:*****************************
  0.05   285:*******************************************
  0.08   332:**************************************************
  0.10   322:************************************************
  0.13   289:********************************************
  0.15   285:*******************************************
  0.18   223:**********************************
  0.20   175:**************************
  0.23   190:*****************************
  0.25   175:**************************
  0.28   152:***********************
  0.30   122:******************
  0.33    95:**************
  0.35    82:************
  0.38    54:********
  0.40    36:*****
  0.43    32:*****
  0.45    42:******
  0.48    24:****
  0.50    14:**
  0.53    10:**
  0.55     2:
  0.58     2:
  0.60     0:
  0.63     0:
  0.65     0:
  0.68     0:
  0.70     0:
  0.73     0:
  0.75     0:
  0.78     0:
  0.80     0:
  0.83     0:
  0.85     0:
  0.88     0:
  0.90     0:
  0.93     0:
  0.95     0:
  0.98     0:
  1.00     0:
```

Figure 3: Illustrating the effect of time of day and pre or post playback on detection probability for GCWA.

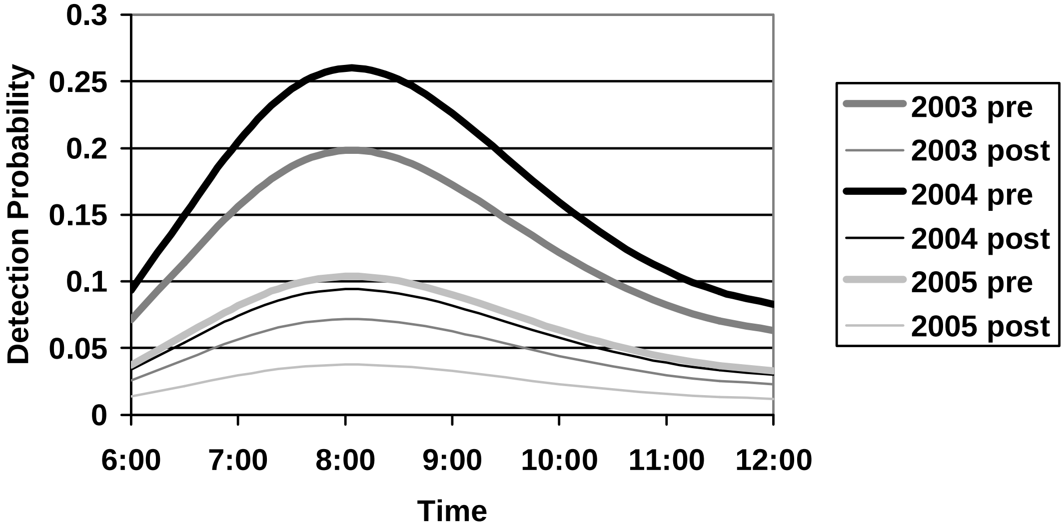
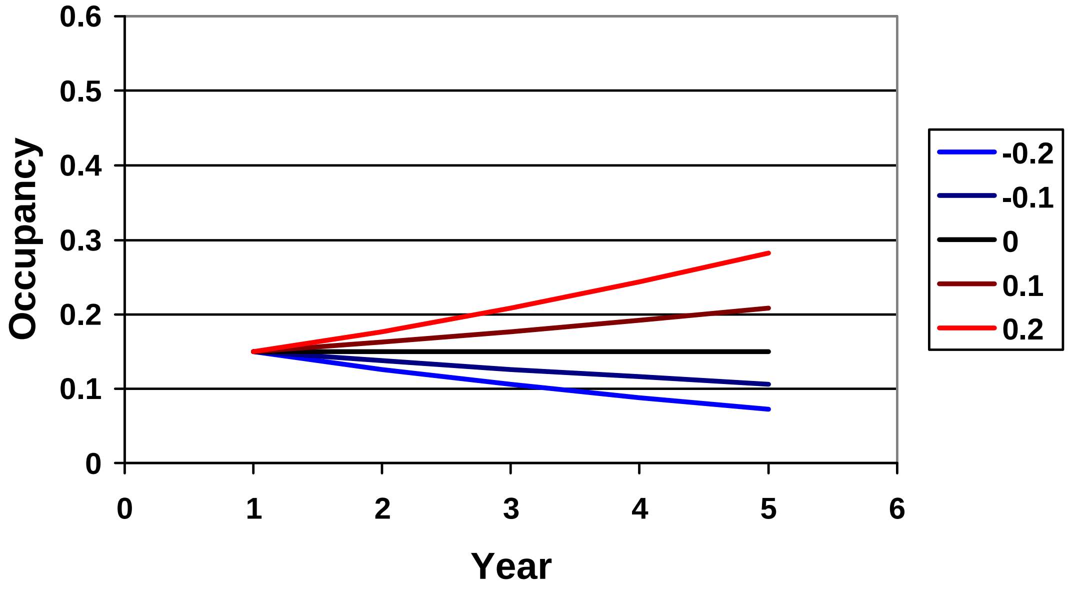


Figure 4: Histogram of the estimated detection probabilities for all surveys conducted from 2003-2005 for BCVI from PRESENCE.

```
Distribution of p's:
  0.00    12:*
  0.03   113:************
  0.05   405:******************************************
  0.08   478:**************************************************
  0.10   477:**************************************************
  0.13   386:****************************************
  0.15   307:********************************
  0.18   260:***************************
  0.20   202:*********************
  0.23   176:******************
  0.25    89:*********
  0.28    71:*******
  0.30    41:****
  0.33    30:***
  0.35    37:****
  0.38    18:**
  0.40    15:**
  0.43    18:**
  0.45    15:**
  0.48     6:*
  0.50     3:
  0.53     0:
  0.55     0:
  0.58     3:
  0.60     0:
  0.63     0:
  0.65     0:
  0.68     0:
  0.70     0:
  0.73     0:
  0.75     0:
  0.78     0:
  0.80     0:
  0.83     0:
  0.85     0:
  0.88     0:
  0.90     0:
  0.93     0:
  0.95     0:
  0.98     0:
  1.00     0:
```

Figure 5: Effect of trends (on the logistic scale) of differing magnitudes on probability of occupancy where the initial probability of occupancy is a) 0.15 and b) 0.3.

a)



b)

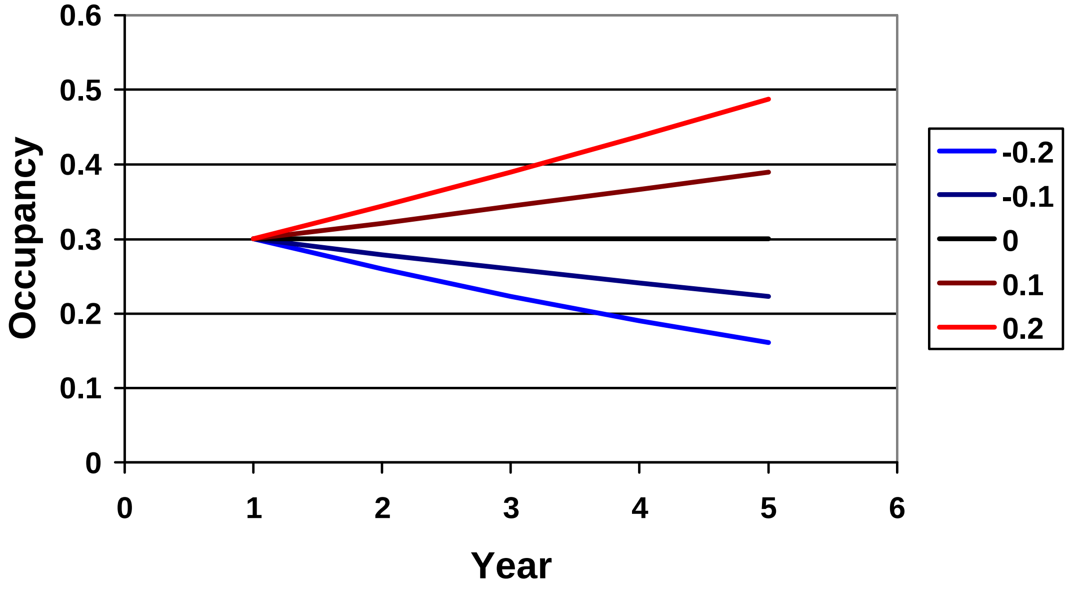
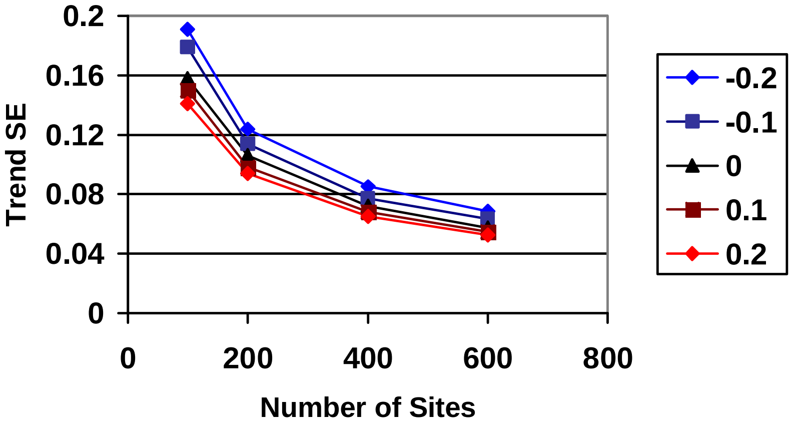
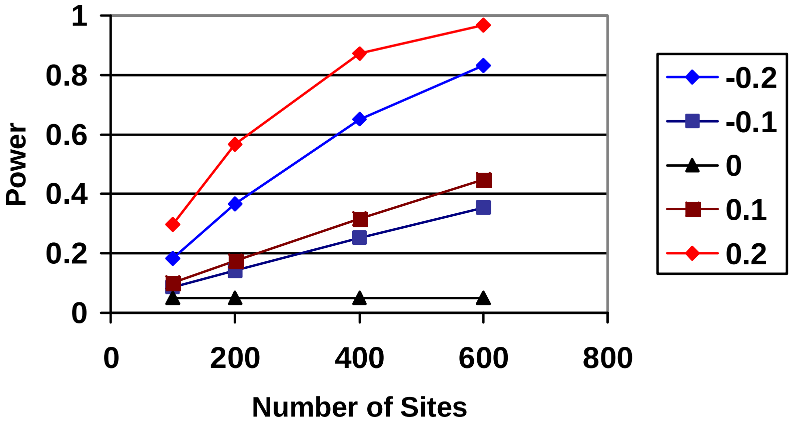


Figure 6: Results from 5000 simulated sets of data with initial level of occupancy = 0.15, *p*=0.2, and 5 surveys per site per year for 5 years. Presented are the approximated a) standard error, b) power with α = 5%, and c) power with α = 10%.

a)



b)



c)

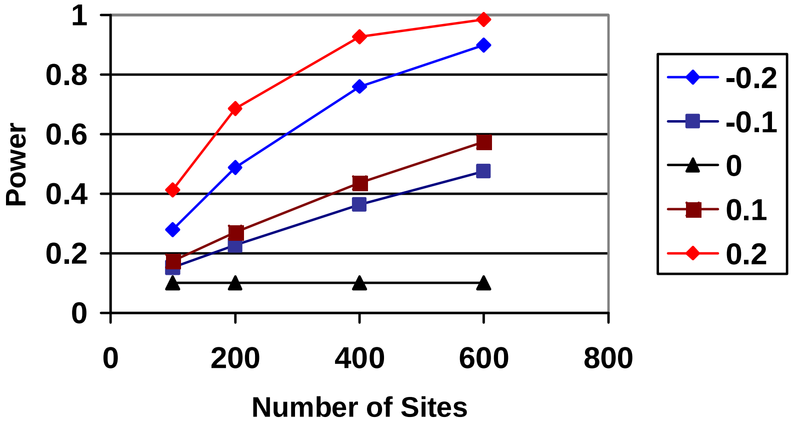
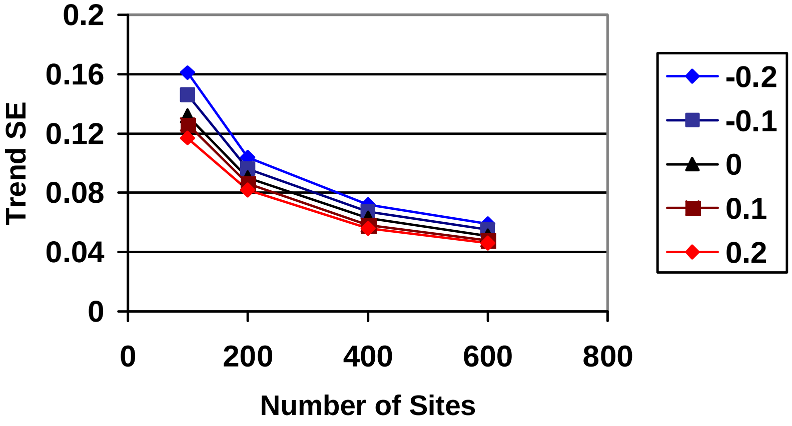
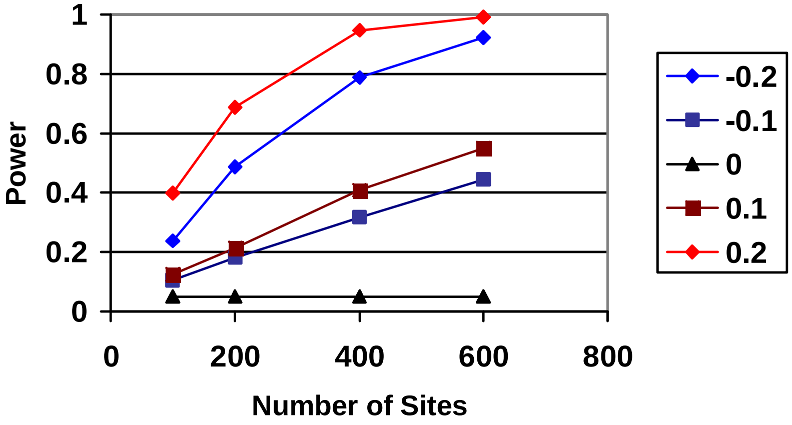


Figure 7: Results from 5000 simulated sets of data with initial level of occupancy = 0.15, *p*=0.2, and 6 surveys per site per year for 5 years. Presented are the approximated a) standard error, b) power with α = 5%, and c) power with α = 10%.

a)



b)



c)

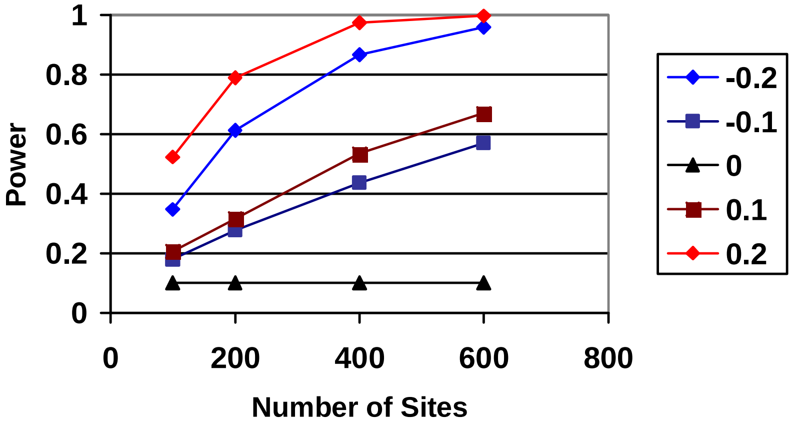
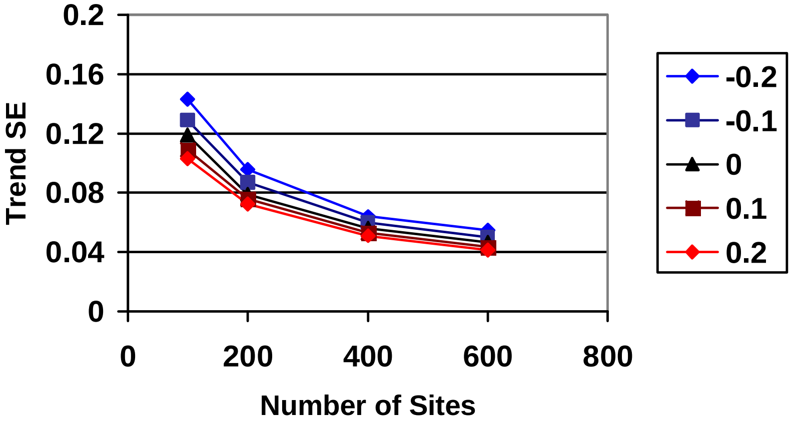
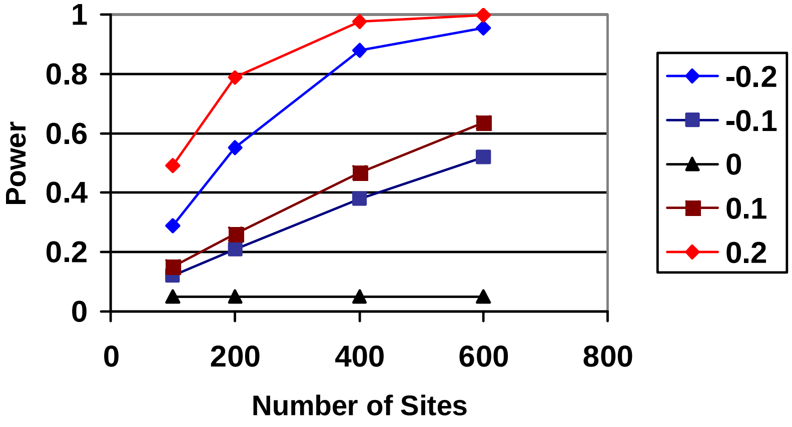


Figure 8: Results from 5000 simulated sets of data with initial level of occupancy = 0.15, *p*=0.2, and 7 surveys per site per year for 5 years. Presented are the approximated a) standard error, b) power with α = 5%, and c) power with α = 10%.

a)



b)



c)

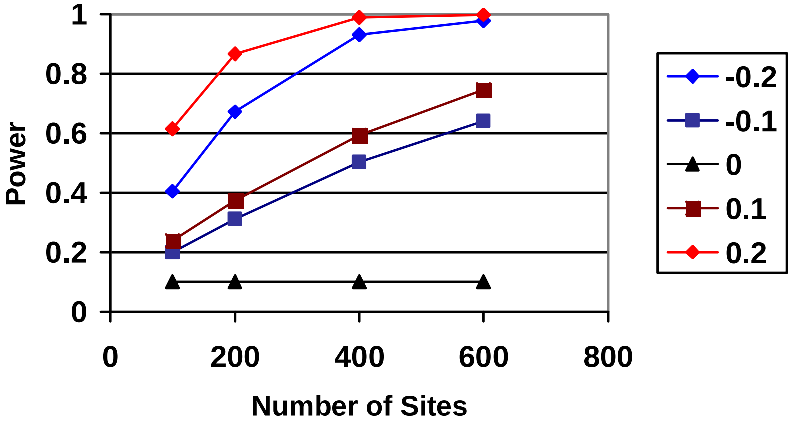
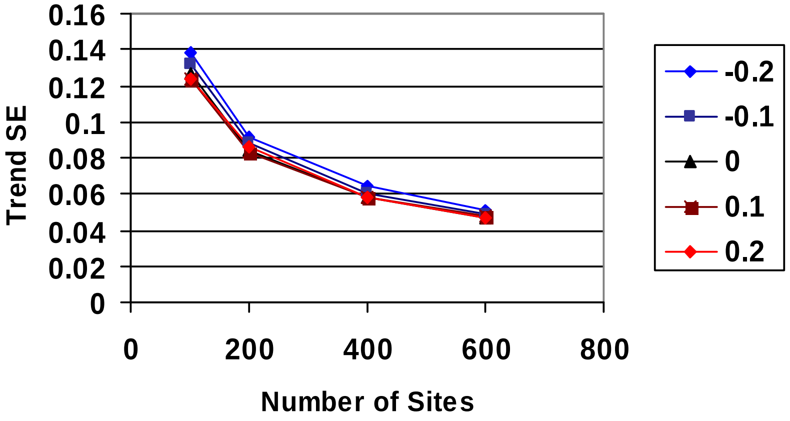
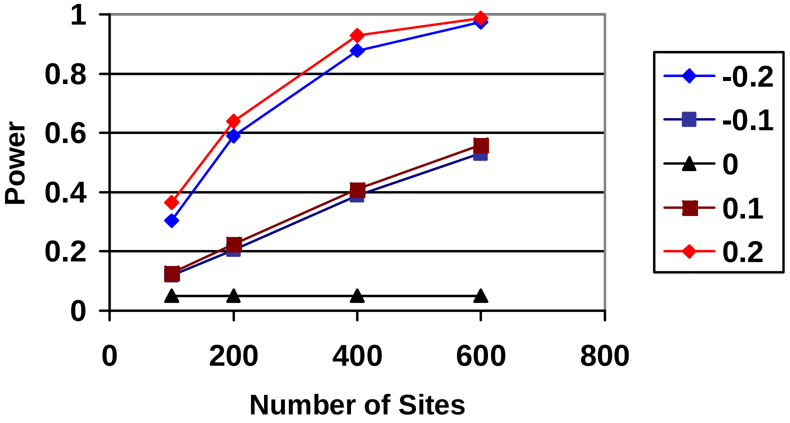


Figure 9: Results from 5000 simulated sets of data with initial level of occupancy = 0.3, *p*=0.2, and 5 surveys per site per year for 5 years. Presented are the approximated a) standard error, b) power with α = 5%, and c) power with α = 10%.

a)



b)



c)

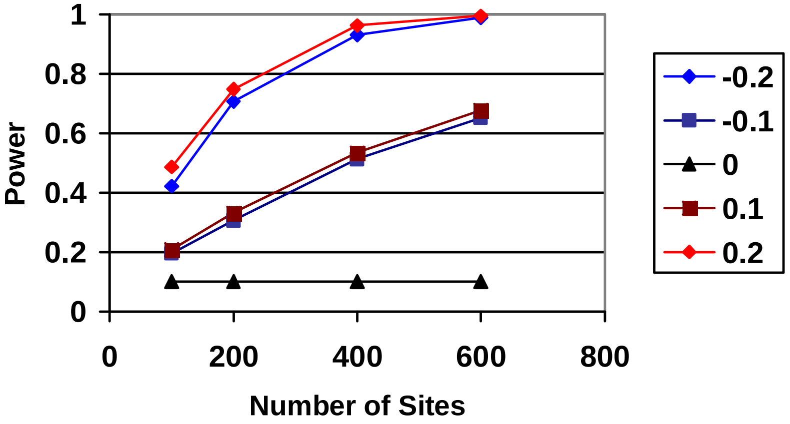
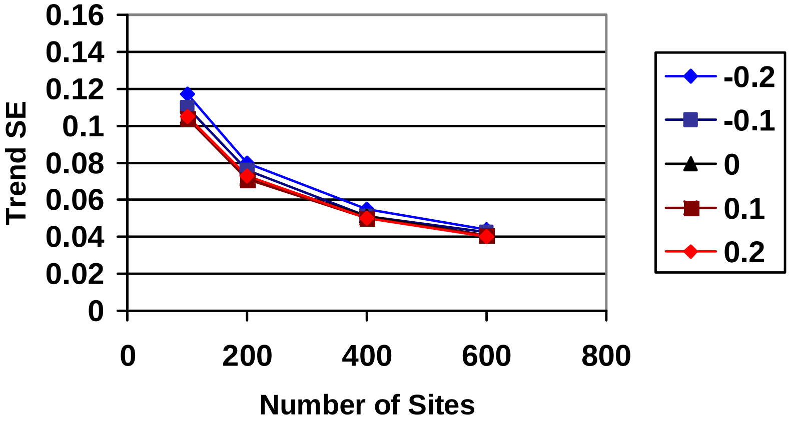
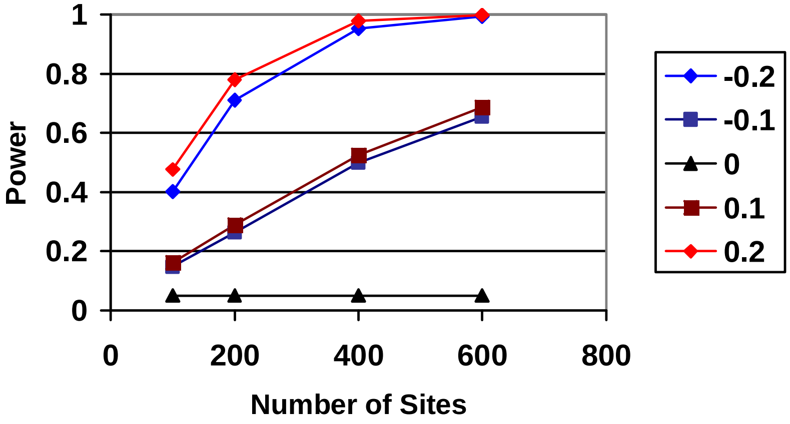


Figure 10: Results from 5000 simulated sets of data with initial level of occupancy = 0.3, *p*=0.2, and 6 surveys per site per year for 5 years. Presented are the approximated a) standard error, b) power with α = 5%, and c) power with α = 10%.

a)



b)



c)

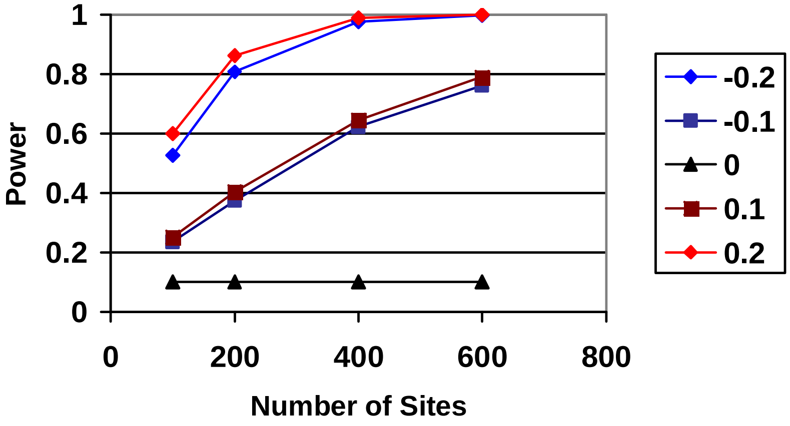
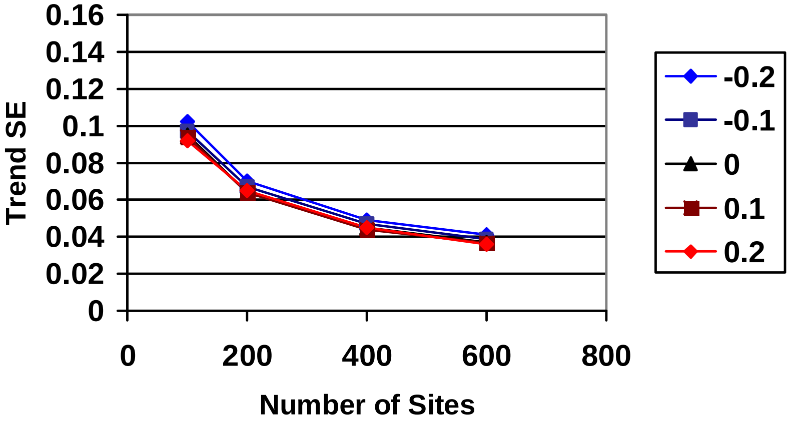
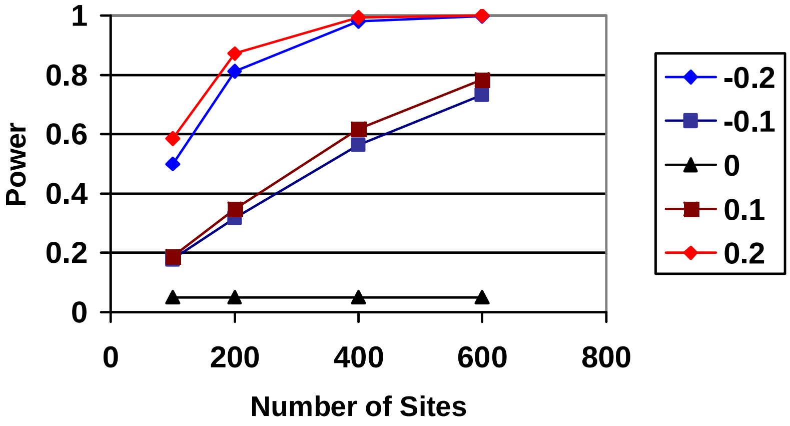


Figure 11: Results from 5000 simulated sets of data with initial level of occupancy = 0.3, *p*=0.2, and 7 surveys per site per year for 5 years. Presented are the approximated a) standard error, b) power with α = 5%, and c) power with α = 10%.

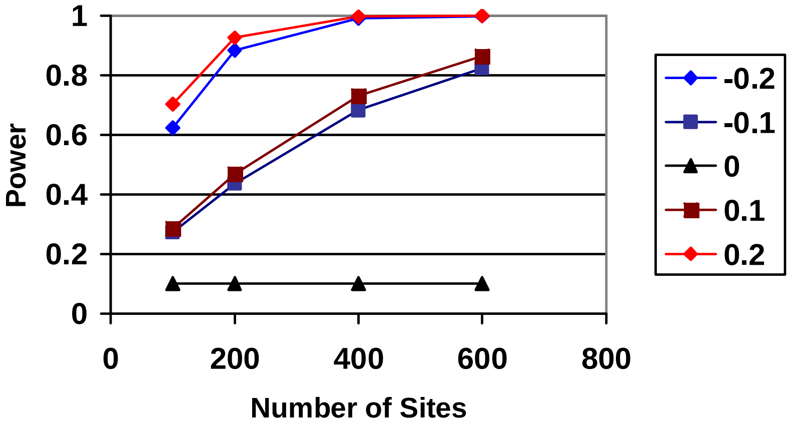
a)



b)



c)



Appendix B

Analysis of 2003-2006 occupancy data for black-capped vireo and golden-cheeked warbler

Analysis of 2003-2006 Occupancy Data for Black-capped Vireo and Golden-cheeked Warbler

Darryl I. MacKenzie

2 April 2007

Proteus
wildlife research consultants

Table of Contents

Executive Summary

- For GCWA:
    - Occupancy increases with percent slope over the range 0-40%.
    - No evidence of a systematic trend in occupancy over time, but there is evidence of annual variation.
    - Detectability increases with percent total canopy cover.
    - Within 2006, detectability decreases though the season.
    - If detected pre-playback then GCWA is more likely to be detected post-playback.
- For BCVI:
    - Data was very sparse limiting the reliability of the AIC model selection
    - Occupancy appears to be decreasing across the study area.
    - Detectability increases with percent total canopy cover.
    - Within 2006, detectability was low in the first 2 surveys and higher in the latter surveys.
    - If detected pre-playback then BCVI is more likely to be detected post-playback.
- It must be noted that apparent temporal changes from the modeling may be real, but may also be the result of not surveying the same sites annually, i.e. spatial and temporal changes in occupancy may be confounded.
- Design recommendations:
    - Continue to conduct 6 surveys per site to overlap the high detectability periods for GCWA (early-mid season) and BCVI (mid-late season).
    - Only use playback if target species is not detected in the initial surveying period.
    - Survey the same sites each year.
    - Survey at least 200 sites per annum in order to have moderate power to detect a moderate trend in occupancy within 5 years.
- Consider very carefully how the monitoring program should be integrated into the desired management and conservation goals for GCWA and BCVI. Focusing on ‘trend detection’ may not be the most useful use of resources or the most relevant information required.

Introduction

Data collected near Fort Hood from 2003-2006 on golden-cheeked warblers (GCWA) and black-capped vireos (BCVI) had been analyzed previously to investigate what factors appeared to be affecting detection probabilities. The main purpose of previously focusing on detection probabilities was to make recommendations about possible adjustments to the field protocols for the 2006 season. In this report we now consider the factors that may be influencing the probability of occupancy by GCWA and BCVI across the landscape, with particular emphasis on temporal patterns. In addition, to assess the effectiveness of changes made to field protocols in 2006 further analyses were performed on just the 2006 detection data.

It must be noted that because of the relatively short timeframe over which data has been collected, and that the same stations have not been consistently monitored each year, the following results must be interpreted with some caution, particularly with respect to temporal effects on occupancy.

Methods

As in the previous report, due to relatively few stations being monitored in consecutive years the multi-season model of MacKenzie et al. (2003) has been used here with the constraint that changes in occupancy occurred at random (MacKenzie et al. 2006). MacKenzie et al. (2006) note that this constraint is equivalent to modeling only the patterns of occupancy each year and not the dynamic processes themselves.

Following discussions with Michael Morrison and Bret Collier, sets of candidate models were developed for the occupancy and detection probabilities for GCWA and BCVI (Table 1). Models denoted with an interaction term between two factors (e.g., A*B) implies that the main effects for those factors are also present. For occurrence probabilities, 3 factors where considered; 1) whether the station was located in the Coryell watershed (CW); 2) percent total canopy cover (%C); and 3) percent slope (%S). In addition, to assess temporal changes in occupancy, a time factor was included with three levels; 1) year-specific (Yr); 2) a logit-linear trend (Tr; i.e., a linear trend in occupancy on the logit scale); or 3) constant. Detection probabilities were always considered to be year specific (Yr), and the effects of the following 4 factors were considered; 1) time since sunrise (TSS, modeled as a 3^{rd} order polynomial); 2) CW; 3) %C and 4) %S. The percent total canopy cover covariate was standardized such that a transformed value of 0 related to 75% cover and converted onto the decile scale (i.e., divided by 10). The percent slope covariate was also transformed on the decile scale. As *Time* is a 3-level factor, 81 models (3×9×3) were fit to the 2003-2006 data for GCWA and BCVI. Models were ranked according to AIC and AIC model weights were calculated. AIC model weights were summed to summarize the relative importance of each factor on occurrence and detection probabilities.

To address future study design questions, further modeling of only the 2006 data was conducted. The relevant questions of interest was the degree of within season variation in detectability, whether the number of repeated surveys conducted in 2006 (6 per site) was appropriate and whether the use of playback increased overall detection probabilities. As detection probability was the main focus of these questions a general model for occupancy probability was maintained throughout (Yr + CW*%S + %C*%S). To address the issue of within season variation a number of models were suggested (Table 2). From these results, previous recommendations on the required level of survey effort based upon the work of MacKenzie and Royle (2005) were reassessed. Exploratory analyses of the 2003-2005 data (MacKenzie 2006) suggested that perhaps detection probability may have been lower post-playback, however subsequent discussion highlighted that playbacks to elicit calls from GCWA and BCVI were not always conducted and nor where they always recorded. In 2006 the field protocol always required playback to be used and that the data be consistently recorded as to whether detection occurred pore- or post-playback. To assess the effect of playback the 2006 data was analyzed at a finer scale where a ‘survey’ consisted of the pre- or post-playback session. Hence rather than having 6 surveys per site the data was entered into PRESENCE as 12 surveys. A general model was again maintained for occupancy probability, and detection probability was modeled as occasion specific, with an additive post-survey effect. Due to concerns about the potential lack of independence of detections pre- and post-playback an indicator covariate was defined that equaled 1 for the post-playback survey if the species was detected in the pre-playback survey on that day, and 0 otherwise. The intent of this covariate is to allow the equivalent of a ‘trap-happy’ effect on the detection of each species.

Results

Golden-cheeked Warbler

Table 3 presents the top 10 models from the model selection procedure for the GCWA detection data. A consistent feature of these top-ranked models is the inclusion of the Coryell watershed and percent slope factors for occupancy probabilities; where these factors are additive, or possibly an interaction between these factors (so the effect of slope on occupancy is different inside and outside of the Coryell watershed). From the entire set of candidate models that have been considered here, the summed AIC model weights for models that have year-specific occupancy, a logit-linear trend in occupancy or constant occupancy are 0.88, 0.06 and 0.06 respectively. This suggests there is no evidence of a systematic trend in occupancy from these 4 years of data. The 4 top-ranked models all have a similar level of support based upon AIC. These models represent combinations of whether the effect of slope on occupancy is different inside and outside of the Coryell watershed, and whether there is an interaction (or not) between canopy cover and slope on detectability. The estimated effect sizes with respect to the occupancy-related parameters are very similar regardless of the exact structure used to model detection probabilities. Hence here we focus on the two models with a more general structure for detection probabilities, i.e., $\psi(Yr+CW+\%S)p(Yr+\%C*\%S)$ and $\psi(Yr+CW*\%S)p(Yr+\%C*\%S)$. The estimated occupancy-related effect sizes from these models are given in Table 4 and the estimated occupancy probabilities for within the Coryell watershed are illustrated in Figures 1 and 2.

For detection probability, the percent total canopy cover factor is included in all of the top ranked models and often the models where detection probability is a function of only canopy cover or there is an interaction effect between canopy cover and percent slope have very similar ΔAIC values. Recall that models were detection probability was a function of only percent slope or only the main effects of canopy cover and slope were not included in the candidate model set. Models that included the 3rd-order polynomial for time since sunrise as a factor ranked relatively low, with the top ranked model of this type having a ΔAIC value of approximately 37. Although again note that model that included both time since sunrise and canopy cover or percent slope were not included in the candidate set of models. Focusing once again on the top 4 models, the estimated detection-related effect sizes are relatively robust to the exact structure used to model occupancy probabilities. Hence final inferences are based upon the models with a more general occupancy probability structure, i.e., ψ(Yr+CW*%S)*p*(Yr+%C) and ψ(Yr+CW*%S)*p*(Yr+%C*%S). Estimates from both models are given in Table 5 and estimated probabilities in Figure 3 and 4.

When only considering the 2006 detection data for GCWA, the top ranked models (Table 6) consistently suggest that detection probability for GCWA is higher earlier in the season. Time since sunrise appear to be less important than in previous analyses, possibly because the times during which surveying was conducted in 2006 was more limited than in previous years. The estimated detection probabilities from the models *p*(Trend) and *p*(*t*) are illustrated in Figure 5, along with the cumulative detection probability (probability of detecting CGWA at least once after *x* surveys). From the p(.) model detection probability in 2006 was estimated to be 0.34 which is higher than previous analyses have indicated (MacKenzie 2006). Based upon the work of MacKenzie and Royle (2005), the current protocol of 6 surveys per site would seem to be an appropriate average number of surveys per site. Following the suggestions of MacKenzie et al. (2006), if desired, one could have unequal surveying effort across different sites based upon the expected occupancy and detection probabilities calculated from the associated site-specific covariates.

Finally, there is very clear evidence of a 'trap-happy' effect (Table 7) such that if GCWA are detected in the pre-playback period the odds of detection in the post-playback period are 6.96 (95% CI: 4.32 – 11.13) times higher than if GCWA were not detected pre-playback. This lack of independence (detection probability post-playback is different when GCWA are detected/nondetected pre-playback) would suggest that under the current modeling approaches little additional useful information is gained by using playbacks after GCWA have already been detected. When GCWA have not been detected pre-playback, then the odds of detecting them post-playback are decreased by 0.28 (0.19 – 0.41). That is, under current protocols when GCWA have not been detected pre-playback, detection probability in the post-playback period is lower than pre-playback.

Black-capped Vireo

For the BCVI, model selection on the basis of AIC is less clear cut. Table 8 presents the top 15 models as ranked by AIC and note that the AIC model weights are spread amongst a large number of models. However the estimates provided by the first model are biologically implausible, particularly for the years 2003-2005; occupancy is commonly estimated to be >0.50. Furthermore, note that from Table 8, often the negative log-likelihood value is relatively unchanged by the addition of extra parameters, hence the additional parameters are not really providing a better fit to the data and the more complicated models are being ranked close to simpler model by virtue of having a similar number of parameters. This is undoubtedly a consequence of having a very sparse data set (80 detections from 4410 surveys over the 4 year period). Therefore, for BCVI we only focus on the top-ranked model that provided biologically plausible estimates; ψ(Trend+CW)p(Yr+%C) (Table 9). From this model, the odds of occupancy are higher within the Coryell watershed by a factor of 2.61 (1.26 – 5.41) and each year the odds of occupancy decrease by 0.57 (0.32 – 1.00). The estimated occupancy probabilities from this model for sites within and outside the Coryell watershed are illustrated in Figure 6.

There is some indication that detection probability increases with the level of total canopy (odds ratio = 1.12; 0.98 – 1.27) and detection probability was higher in 2006 than previous years (Figure 7).

Focusing on only the 2006 data, time since sunrise appears to be an important covariate for detection probability for BCVI, unlike for the GCWA (Table 10). However, looking at the estimated detection probabilities from the top-ranked model across a range of observed time since sunrise values (Figure 8) does raise some concerns that perhaps, as a result of the small number of detections, a few detections later in the day may be over emphasising the importance of this covariate in the modeling. Therefore in the estimated detection probabilities for the top ranked model without the TSS covariate is also presented (Figure 9).

Analysing the 2006 data at the finer resolution of pre- and post-playback provides a strong indication of a lack of independence in detecting BCVI post-playback based upon the AIC model weights (Table 11). While both of the top 2 ranked models indicate a ‘trap-happy’ effect, the exact magnitude of the effect is uncertain (estimated odds ratios are 8.10 (95% CI: 1.72-37.80) and 3.90 (1.08-14.00) respectively). There is also some indication that detection probability may be lower post-playback when BCVI was not detected pre-playback, but it is not conclusive as the post-playback factor appears in the top and bottom ranked models.

Discussion and Recommendations

Generally the above results support the changes in field protocols that were implemented in the 2006 field season. For both GCWA and BCVI the detection probably (per survey) in 2006 was generally higher than in previous years which may be the result of focusing survey effort on a smaller time interval between sunrise and mid-morning. However estimated detection probabilities were still relatively low and while individually for each species it be possible to use fewer than 6 surveys per site, as there was some indication that GCWA where more detectably early season, and BCVI more detectable later in the season it would seem prudent to continue with 6 surveys per site to encompass the periods of higher detectability for both species. As was noted above, one option could be to tailor the number of repeated surveys somewhat based upon the predicted occupancy and detection probabilities of specific sites, though when sampling for multiple species it may be more convenient to keep the study design simple and survey all sites 6 times.

Both GCWA and BCVI provided strong indications of a ‘trap-happy’ effect when using the pre- and post-playback detection data. This suggests that little additional information (for estimating occupancy) is garnered by using playback if the species has already been detected. For both species there was some indication that detection probability was lower post-playback , although this may partially be a consequence of the shorter time interval used post-playback to hear a response from nearby members of the species. In addition, as detectability of these species appears to be relatively low, efforts should be made at every opportunity to increase detection probability. It is therefore recommended that playbacks continue to be used, but only if the species is not detected in the first 6-minute survey period. Whether playback has been used during a survey should be record and the outcome of the playback survey recorded separately from the results of the initial survey.

As noted above, while the observed trend in occupancy for BCVI and apparent lack of a trend for GCWA may be true, it must be kept in mind that the spatial extent of the sites surveyed from 2003-2006 has not been consistent; the apparent temporal changes in occupancy may be partially a result of surveying a different set of points in the landscape each year. Therefore results regarding temporal changes in occupancy for BCVI and GCWA must be treated with some caution. For the purpose of estimating trends in occupancy across the landscape it is recommended that the same sites are surveyed each year, provided the initial selection of sites could be considered as representative of the wider region that conclusions are to be generalized to.

With regards to trend detection, these latest results do not substantially alter the advice given previously and the power calculations provided in MacKenzie (2006) are still relevant. I would recommend continuing to survey at least 200 sites per year, and more if resources allow. However I would like to raise the point that sometimes focusing on trend detection is not a very useful exercise in the context of attempting to manage wildlife populations. In many situations the available resources will only permit a relatively imprecise estimate of trend over the time required by managers, and it’s worth noting that statisticians that work primarily with time-series applications would consider 20 times points “not a bad start.” Often by the time a trend is deemed ‘significant’ it may be too late for the population in question. Nichols and Williams (2006) have recently argued that in many conservation settings it makes sense to take a more proactive approach to management and realizing that monitoring should not be viewed as a stand alone activity. I strongly recommend that careful thought should be given to how monitoring can be best used to met the goals of GCWA and BCVI conservation.

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Tables

Table 1: Candidate models for occurrence and detection probabilities. The factor *Time* represents either year-specific (Yr), logit-linear trend (Tr) or a constant level of occupancy. The other factors are Coryell Watershed (CW), percent slope (%S), percent total canopy cover (%C) and time since sunrise (TSS).

Model	Occurrence Probability (ψ)	Detection Probability (p)
1	Time	Yr + TSS
2	Time + CW	Yr + %C
3	Time + %C	Yr + %C * %S
4	Time + %S	
5	Time + CW + %S	
6	Time + CW * %S	
7	Time + CW + %C	
8	Time + %C + %S	
9	Time + %C * %S	

Table 2: Candidate models to assess the degree of within season variation in detection probability in 2006. For each of the models listed below, the time since sunrise covariate (TSS) is included and excluded. Numbers in the model descriptions denote survey days with parentheses indicating survey days that are estimated with a common detection probability, i.e., (1,2) (3-6) indicates survey days 1 & 2 have the same detection probability and days 3-6 have a second common detection probability.

Model	Description
1	Constant
2	Survey Day Specific
3	Logit-linear trend
4	(1,2) (3,4) (5,6)
5	(1,2) (3-6)
6	(1-3) (4-6)
7	(1-4) (5,6)
8	(1,6) (2-5)
9	(1,6) (2,3) (4,5)

Table 3: Top 10 AIC-ranked models for 2003-2006 GCWA detection data. The candidate model set was defined as per Table1. ΔAIC is the difference in AIC values between the top-ranked and present model, *w* is the AIC model weight, CS *w* is the cumulative sum of AIC model weights, *K* is the number of parameters and −2*l* is twice the negative log-likelihood value.

Model	ΔAIC	*w*	CS *w*	*K*	−2*l*
ψ(Yr+CW+%S)*p*(Yr+%C)	0.00	0.26	0.26	11	2328.25
ψ(Yr+CW+%S)*p*(Yr+%C*%S)	0.26	0.23	0.48	13	2324.51
ψ(Yr+CW*%S)*p*(Yr+%C)	0.30	0.22	0.70	12	2326.55
ψ(Yr+CW*%S)*p*(Yr+%C*%S)	0.73	0.18	0.88	14	2322.98
ψ(CW*%S)*p*(Yr+%C)	5.20	0.02	0.90	9	2337.45
ψ(CW*%S)*p*(Yr+%C*%S)	5.47	0.02	0.92	11	2333.72
ψ(Trend+CW*%S)*p*(Yr+%C)	5.59	0.02	0.93	10	2335.84
ψ(Trend+CW+%S)*p*(Yr+%C)	5.66	0.02	0.95	9	2337.91
ψ(CW+%S)*p*(Yr+%C*%S)	5.76	0.01	0.96	10	2336.01
ψ(CW+%S)*p*(Yr+%C)	5.79	0.01	0.98	8	2340.04

Table 4: Parameter estimates for the highly ranked occupancy models with a general structure for detection probabilities, *p*(Yr+%C*%S), for GCWA.

	ψ(Yr+CW*%S)		ψ(Yr+CW+%S)	
Factor	Estimate	SE	Estimate	SE
2003	-1.80	0.50	-1.98	0.50
2004	-2.87	0.51	-3.13	0.47
2005	-3.10	0.57	-3.29	0.57
2006	-1.79	0.37	-2.01	0.34
CW	1.20	0.55	1.75	0.32
%S	0.70	0.22	0.86	0.20
CW*%S	0.42	0.35	-	-

Table 5: Parameter estimates for the highly ranked detection model with a general structure for occupancy probabilities, ψ(Yr+CW*%S), for GCWA

	p(Yr+%C*%S)		*p*(Yr+%C)	
Factor	Estimate	SE	Estimate	SE
2003	-1.93	0.33	-1.85	0.24
2004	-1.28	0.24	-1.27	0.18
2005	-1.90	0.38	-1.83	0.32
2006	-1.04	0.20	-1.03	0.12
%C	0.15	0.07	0.25	0.03
%S	0.03	0.09	-	-
%C*%S	0.06	0.04	-	-

Table 6: Model selection summary for 2006 GCWA detection data. The candidate model set for detection probability was defined as per Table2. ΔAIC is the difference in AIC values between the top-ranked and present model, *w* is the AIC model weight, CS *w* is the cumulative sum of AIC model weights, *K* is the number of parameters and $-2l$ is twice the negative log-likelihood value. For all models the a general structure was maintained for the occupancy component of the model, ψ(Yr + CW*%S + %C*%S).

Model	ΔAIC	*w*	CS *w*	*K*	$-2l$
p(Trend)	0.00	0.60	0.60	8	855.24
p(Trend+TSS)	1.70	0.26	0.85	11	850.94
p(*t*)	4.79	0.05	0.91	12	852.03
p(*t*+TSS)	5.83	0.03	0.94	15	847.07
p((1,2) (3,4) (5,6))	6.78	0.02	0.96	9	860.02
p((1,2) (3-6))	7.79	0.01	0.97	8	863.03
p((1-3) (4-6))	8.56	0.01	0.98	8	863.80
p((1,2) (3,4) (5,6)+TSS)	8.90	0.01	0.99	12	856.14
p((1,2) (3-6)+TSS)	9.57	0.01	0.99	11	858.81
p((1-4) (5,6))	9.96	0.00	1.00	8	865.20
p((1-3) (4-6)+TSS)	10.56	0.00	1.00	11	859.80
p((1-4) (5,6)+TSS)	12.69	0.00	1.00	11	861.93
p(.)	18.93	0.00	1.00	7	876.18
p((1,6) (2-5))	20.53	0.00	1.00	8	875.78
p(TSS)	21.34	0.00	1.00	10	872.58
p((1,6) (2,3) (4,5))	21.66	0.00	1.00	9	874.90
p((1,6) (2-5)+TSS)	22.83	0.00	1.00	11	872.07
p((1,6) (2,3) (4,5)+TSS)	23.97	0.00	1.00	12	871.21

Table 7: Model selection summary for 2006 GCWA detection data pre/post playback. ΔAIC is the difference in AIC values between the top-ranked and present model, *w* is the AIC model weight, CS *w* is the cumulative sum of AIC model weights, *K* is the number of parameters and $-2l$ is twice the negative log-likelihood value. For all models the a general structure was maintained for the occupancy component of the model, ψ(Yr + CW*%S + %C*%S). 'day' denotes a different detection probability for each survey day, 'post' represents an additive effect on post-playback detection probability, and 'ind.' allows detection probability in the pos-playback period to be different when GCWA are detected/not detected in the pre-playback period.

Model	ΔAIC	*w*	CS *w*	*K*	$-2l$
p(day + post + ind.)	0.00	1.00	1.00	14	1320.16
p(day + ind.)	48.80	0.00	0.00	13	1370.96
p(day + post)	69.80	0.00	0.00	13	1391.96
p(day)	78.35	0.00	0.00	12	1402.51

Table 8: Top 15 AIC-ranked models for 2003-2006 BCVI detection data. The candidate model set for detection probability was defined as per Table1. ΔAIC is the difference in AIC values between the top-ranked and present model, *w* is the AIC model weight, CS *w* is the cumulative sum of AIC model weights, *K* is the number of parameters and $-2l$ is twice the negative log-likelihood value.

Model	ΔAIC	*w*	CS *w*	*K*	$-2l$
ψ(Yr+%C*%S)p(Yr+%C*%S)	0.00	0.46	0.46	14	717.37
ψ(Trend+CW)p(Yr+%C)	2.27	0.15	0.60	8	731.64
ψ(Trend+CW+%C)p(Yr+%C)	3.90	0.06	0.67	9	731.27
ψ(Trend+CW+%S)p(Yr+%C)	4.27	0.05	0.72	9	731.64
ψ(Trend+CW*%S)p(Yr+%C)	6.02	0.02	0.74	10	731.39
ψ(Yr+CW)p(Yr+%C)	6.02	0.02	0.77	10	731.39
ψ(Trend+CW)p(Yr+%C*%S)	6.04	0.02	0.79	10	731.41
ψ(CW)p(Yr+%C)	6.62	0.02	0.80	7	737.99
ψ(Trend+CW+%S)p(Yr+%C*%S)	6.79	0.02	0.82	11	730.16
ψ(Trend+CW+%C)p(Yr+TSS)	6.79	0.02	0.84	11	730.16
ψ(CW+%C)p(Yr+%C)	7.34	0.01	0.85	8	736.71
ψ(Trend+%C*%S)p(Yr+%C*%S)	7.47	0.01	0.86	12	728.84
ψ(Trend+CW)p(Yr+TSS)	7.48	0.01	0.87	10	732.85
ψ(Trend+CW+%C)p(Yr+%C*%S)	7.62	0.01	0.88	11	730.99
ψ(Trend)p(Yr+%C)	7.69	0.01	0.89	7	739.06
ψ(Yr+%C*%S)p(Yr+%C*%S)	0.00	0.46	0.46	14	717.37
ψ(Trend+CW)p(Yr+%C)	2.27	0.15	0.60	8	731.64
ψ(Trend+CW+%C)p(Yr+%C)	3.90	0.06	0.67	9	731.27

Table 9: Parameter estimates from the highest ranked model that provided biologically plausible estimates for BCVI, ψ(Trend+CW)*p*(Yr+%C).

	ψ(Trend+CW)		*p*(Yr+%C)	
Factor	Estimate	SE	Estimate	SE
2003	-1.80	0.50	-1.98	0.50
2004	-2.87	0.51	-3.13	0.47
2005	-3.10	0.57	-3.29	0.57
2006	-1.79	0.37	-2.01	0.34
CW	1.20	0.55	1.75	0.32
%C	-	-	0.86	0.20

Table 10: Model selection summary for 2006 BCVI detection data. The candidate model set for detection probability was defined as per Table2. ΔAIC is the difference in AIC values between the top-ranked and present model, *w* is the AIC model weight, CS *w* is the cumulative sum of AIC model weights, *K* is the number of parameters and $-2l$ is twice the negative log-likelihood value. For all models the a general structure was maintained for the occupancy component of the model, ψ(Yr + CW*%S + %C*%S).

Model	ΔAIC	*w*	CS *w*	*K*	$-2l$
p((1,2) (3-6) + TSS)	0.00	0.61	0.61	11	127.63
p((1,2) (3,4) (5,6) + TSS)	1.44	0.30	0.90	12	127.07
p(*t* + TSS)	3.90	0.09	0.99	15	123.53
p((1,2) (3-6))	9.82	0.00	0.99	8	143.45
p(Trend + TSS))	10.41	0.00	1.00	11	138.04
p((1,2) (3,4) (5,6))	11.35	0.00	1.00	9	142.98
p((1-3) (4-6) + TSS)	14.25	0.00	1.00	11	141.88
p(*t*)	14.34	0.00	1.00	12	139.97
p((1-3) (4-6))	15.19	0.00	1.00	8	148.82
p(.)	15.26	0.00	1.00	7	150.89
p(TSS)	16.60	0.00	1.00	10	146.23
p((1-4) (5,6))	16.86	0.00	1.00	8	150.49
p((1,6) (2-5))	17.21	0.00	1.00	8	150.84
p((1-4) (5,6) + TSS)	17.61	0.00	1.00	11	145.24
p((1,6) (2-5) + TSS)	18.59	0.00	1.00	11	146.22
p((1,6) (2,3) (4,5)	19.15	0.00	1.00	9	150.78
p((1,6) (2,3) (4,5) + TSS)	20.31	0.00	1.00	12	145.94
p(Trend)	25.33	0.00	1.00	8	158.96

Table 11: Model selection summary for 2006 BCVI detection data pre/post playback. ΔAIC is the difference in AIC values between the top-ranked and present model, *w* is the AIC model weight, CS *w* is the cumulative sum of AIC model weights, *K* is the number of parameters and –2*l* is twice the negative log-likelihood value. For all models the a general structure was maintained for the occupancy component of the model, ψ(Yr + CW*%S + %C*%S). 'day' denotes a different detection probability for each survey day, 'post' represents an additive effect on post-playback detection probability, and 'ind.' allows detection probability in the pos-playback period to be different when GCWA are detected/not detected in the pre-playback period.

Model	ΔAIC	*w*	CS *w*	*K*	–2*l*
p(day + post + ind.)	0.00	0.62	0.62	14	187.45
p(day + ind.)	1.68	0.27	0.89	13	191.13
p(day)	4.23	0.08	0.97	12	195.68
p(day + post)	5.79	0.03	1.00	13	195.24

Figures

Figure 1:Predicted probability of occupancy in Coryell watershed in 2003-2006 for GCWA versus percent slope. Predictions made according to the fitted model ψ(Yr+CW*%S)*p*(Yr+%C*%S).

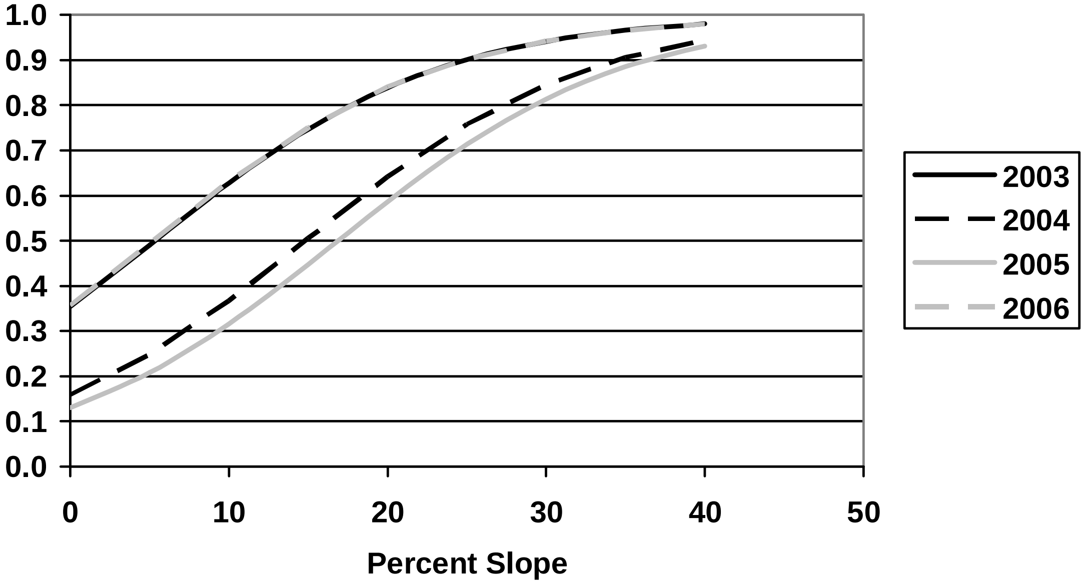


Figure 2: Predicted probability of occupancy in Coryell watershed in 2003-2006 for GCWA versus percent slope. Predictions made according to the fitted model ψ(Yr+CW+%S)*p*(Yr+%C*%S).

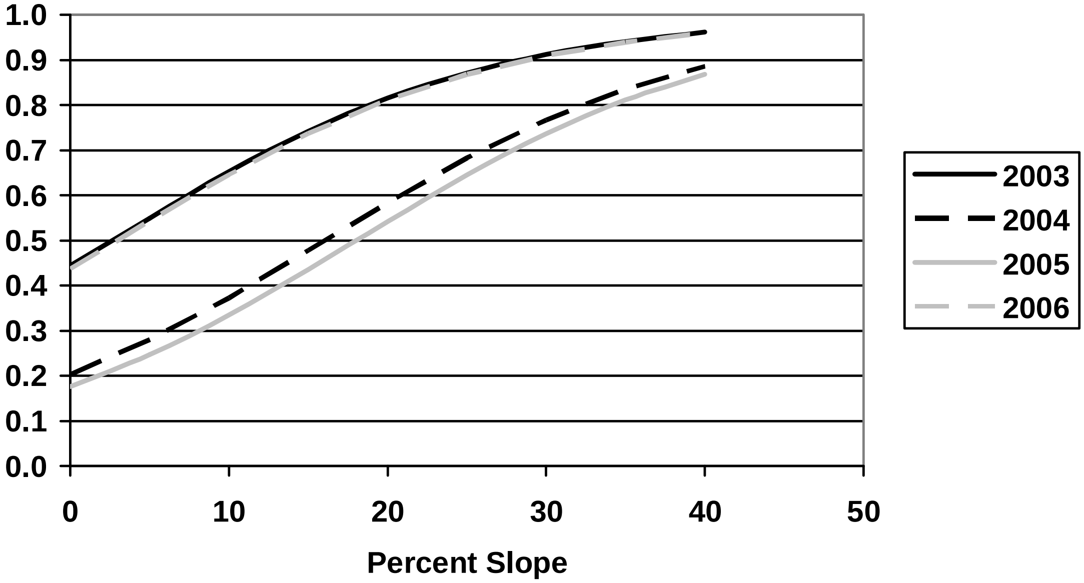


Figure 3: Predicted probability of detection in 2003-2006 for GCWA versus percent total canopy cover. Predictions made according to the fitted model $\psi(Yr+CW*\%S)p(Yr+\%C*\%S)$. The panels illustrate how the effect of canopy changes with percent slope; a) slope = 0%; b) slope = 20%; and c) slope = 40%.

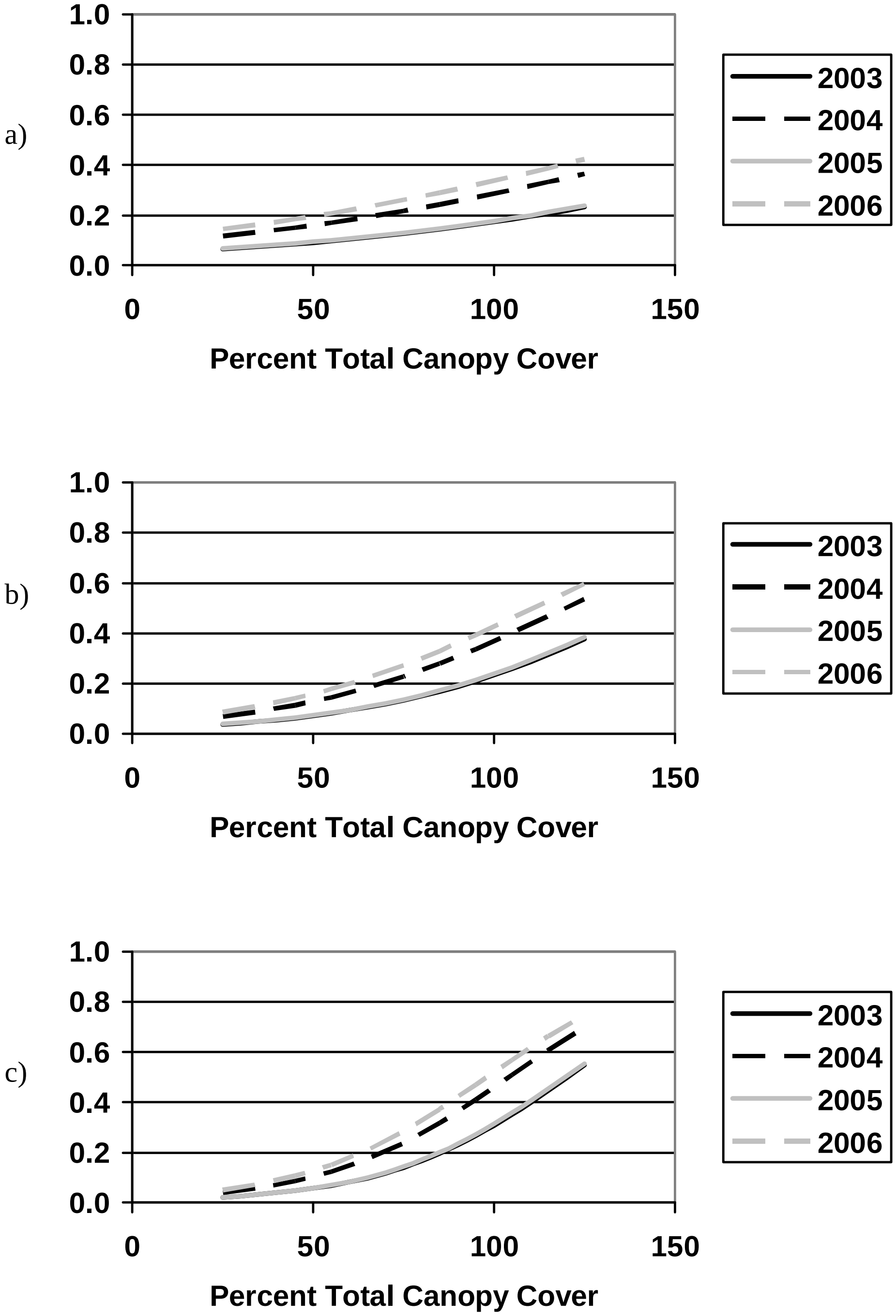


Figure 4: Predicted probability of detection in 2003-2006 for GCWA versus percent total canopy cover. Predictions made according to the fitted model ψ(Yr+CW*%S)*p*(Yr+%C)..

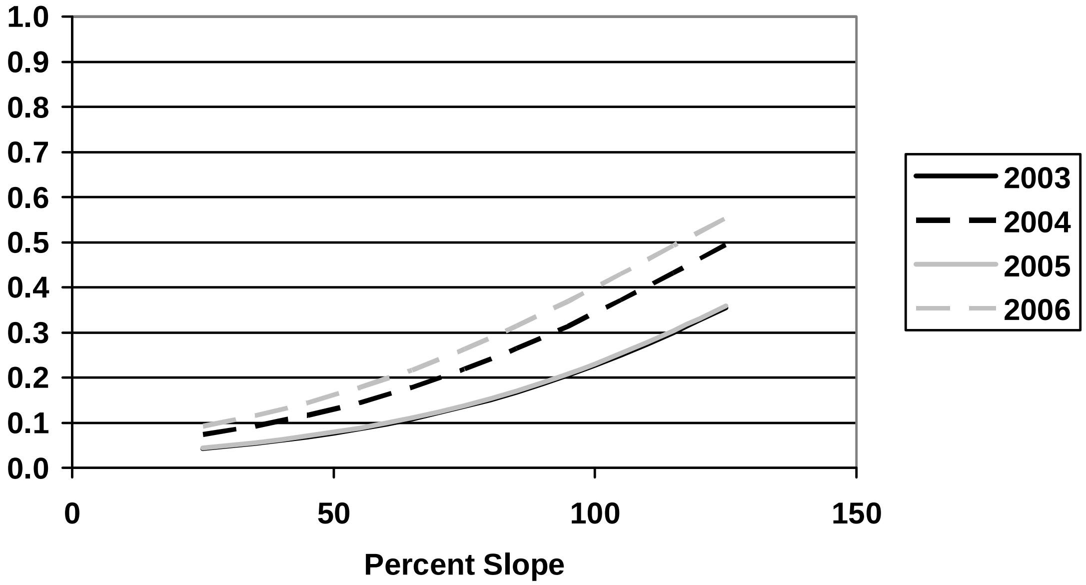


Figure 5: Predicted probability of detecting GCWA in 2006 within a season by survey number. In both models occupancy was modelled as ψ(Yr+CW*%S+%C*%S). Also plotted is the cumulative detection probability for each model (hollow shapes).

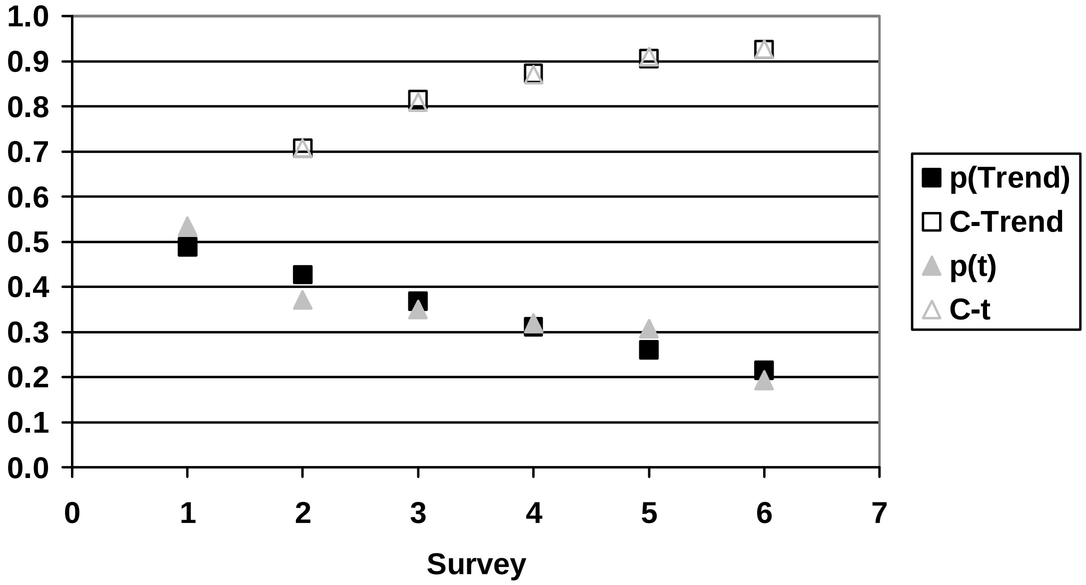


Figure 6: Predicted probability of occupancy for BCVI inside and outside of the Coryell watershed in 2003-2006. Predictions made according to the fitted model $\psi(\text{Trend+CW})p(\text{Yr+\%C})$.

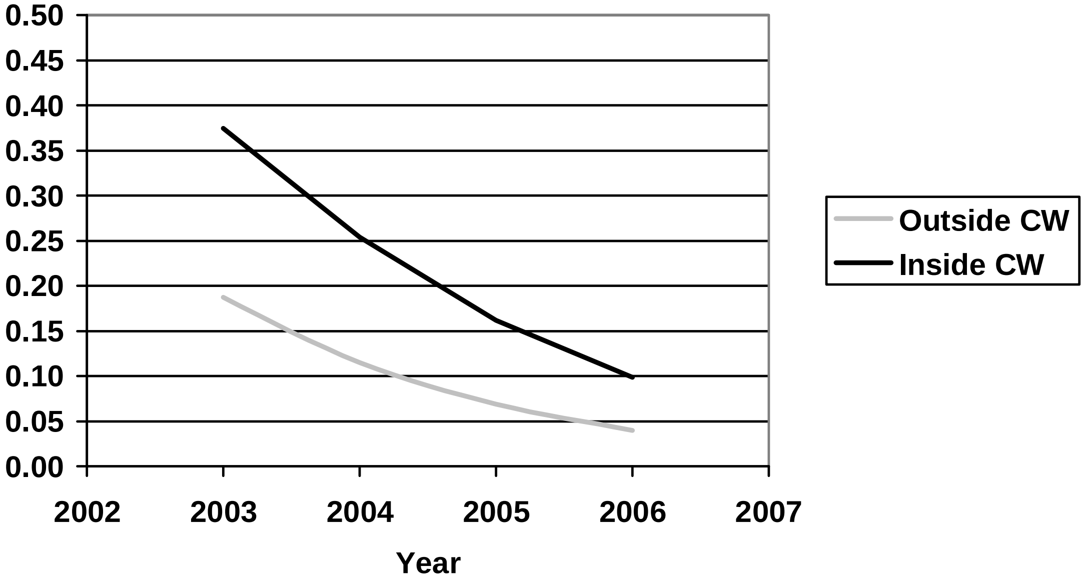


Figure 7: Predicted probability of detection for BCVI versus percent total canopy cover in 2003-2006. Predictions made according to the fitted model $\psi(\text{Trend+CW})p(\text{Yr+\%C})$.

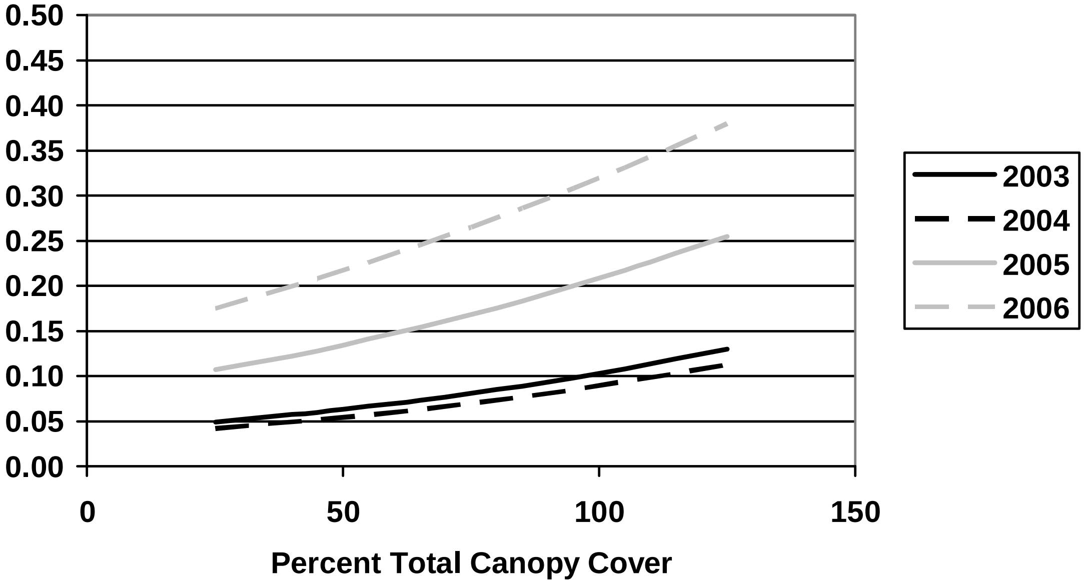


Figure 8: Predicted probability of detecting BCVI in 2006 from the top-ranked model in Table 10. Occupancy was modelled as ψ(Yr+CW*%S+%C*%S). Grey line is for surveys 1 and 2, and black line is for surveys 3-6.

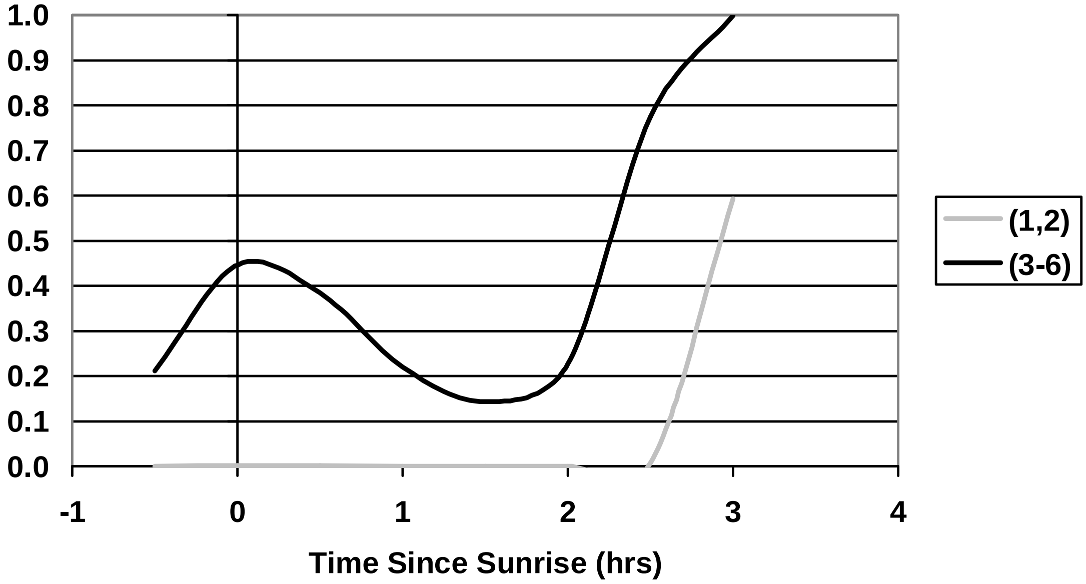
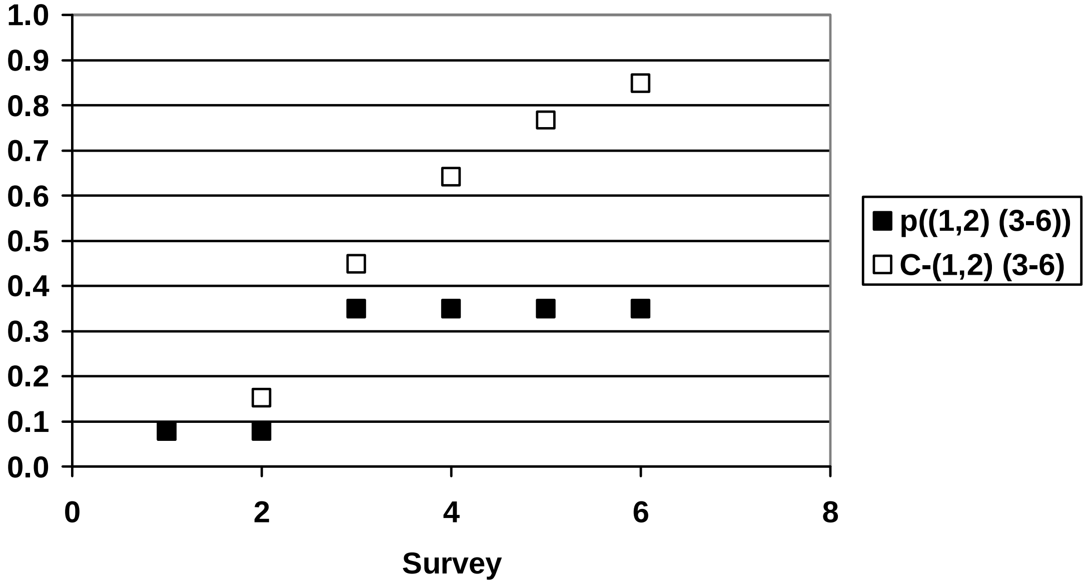


Figure 9: Predicted probability of detecting BCVI in 2006 within a season by survey number for the top-ranked model without the time since sunrise covariate. Occupancy was modelled as ψ(Yr+CW*%S+%C*%S). Also plotted is the cumulative detection probability (hollow square).



Appendix C

Minimum patch size thresholds of reproductive success of songbirds

MINIMUM PATCH SIZE THRESHOLDS OF REPRODUCTIVE SUCCESS OF SONGBIRDS

A Dissertation

by

JERROD ANTHONY BUTCHER

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

May 2008

Major Subject: Wildlife and Fisheries Sciences

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Approved by:

Co-Chairs of Committee,	R. Dean Ransom
	Michael L. Morrison
Committee Members,	R. Douglas Slack
	Fred E. Smeins
Head of Department,	Thomas Lacher, Jr.

May 2008

Major Subject: Wildlife and Fisheries Sciences

ABSTRACT

Minimum Patch Size Thresholds of Reproductive Success of Songbirds.

(May 2008)

Jerrod Anthony Butcher, B.S., Texas A&M University; M.S. University of Texas at Arlington

Co-Chairs of Advisory Committee: Dr. R. Dean Ransom
Dr. Michael L. Morrison

Preservation of large tracts of habitat is often recommended for long-term population viability of area-sensitive species. Large tracts may not always be available. Smaller patches, though not able to contain a viable population individually, may contribute to overall regional population viability if within the small patches pairs could successfully reproduce. By definition, area-sensitive species should have a minimum patch size threshold of habitat below which they will not likely reproduce. Two potential causes for positive relationships between patch size and production are inverse relationships between patch size and brood parasitism and patch size and food availability. My objectives were (1) to determine the minimum patch size thresholds of reproductive success for golden-cheeked warblers (*Dendroica chrysoparia*), black-and-white warblers (*Mniotilta varia*), and white-eyed vireos (*Vireo griseus*); (2) to determine whether thresholds for occupancy, territory establishment by males, or pairing success were indicative of thresholds of reproduction; (3) to determine whether the proportion of pairs fledging brown-headed cowbird (*Molothrus ater*) young was related to patch size, and (4) to determine the affects of patch size on food availability (i.e., arthropod abundance). The Vickery index of reproductive activity was used to determine reproductive activity of each male or pair and to quantify parasitism occurrences. I collected arthropods using branch clipping to assess the relationship between patch size and arthropod abundance. I found minimum patch size thresholds of reproductive success for golden-cheeked and black-and-white warblers, but not for white-eyed vireos. Minimum patch size of reproductive success was between 15 and 20.1 ha. Minimum patch size thresholds for occupancy, territory establishment by males, and pair formation were not consistent with thresholds for reproductive success. I found no relationships between patch size and cowbird parasitism or patch size and arthropod biomass. Conservation practices for target species based on thresholds of occupancy, territory establishment, or pair formation may not address issues of reproduction. The ability to identify thresholds of reproductive success for target species could be useful in conservation and management in multiple ways including setting goals for retention and restoration of a target species' habitat patch size.

ACKNOWLEDGEMENTS

I am grateful to the many people who helped me succeed at this level. Each member of my committee played a unique and needed role. Drs. M. L. Morrison and R. D. Ransom helped me focus on my study topic at a time when I wanted to do too much. Their strategic questions and continuous encouragement helped me maintain focus and improve on preliminary ideas. Drs. R. D. Slack and F. E. Smeins helped me to expand my approach of the study of songbirds to a more encompassing view including the social and historical issues related to conservation and natural history. I am very thankful to my fellow graduate students for their input and discussions about my project. I am especially thankful for the input of S. L. Farrell and A. S. Campomizzi into this study.

Many people at the Institute of Renewable Natural Resources, Texas A&M University, were integral in the logistics of the project. Dr. R. N. Wilkins obtained the funding and pushed me to conduct research that addressed practical management needs. A. Grones and A. Hays were always willing to share their GIS and remote sensing knowledge. K. B. Hays and L. Law made the project happen through addressing many logistic barriers.

I would like to thank the Department of Defense, Environmental Readiness Program; the United States Department of Agriculture, Natural Resources Conservation Service; and Texas Parks and Wildlife Department for funding my research.

Although the above mentioned people were significant in my research and education, I would not have made it this far without my wife, Lourdes Butcher, and my parents, Jerry and Judy Butcher. I am so thankful for their support and constant encouragement.

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INTRODUCTION

It is often recommended that large tracts of habitat be preserved for long-term population viability of area-sensitive species (e.g., Wahl et al. 1990, Donovan et al. 1995, Beardmore et al. 1996). Large tracts may not always be available for preservation particularly in regions where most of the area is privately owned (e.g., Texas, where 94% of land is privately owned [Texas Parks and Wildlife Department 2007]). Smaller patches, though not able to maintain a viable population without emigration from outside sources, may contribute to overall regional population viability if within the small patches pairs could successfully reproduce. The question then is how large of a patch is required for successful breeding by an area-sensitive species. That is, what is the minimum patch size threshold above which reproduction will likely occur?

The concept of thresholds has pervaded ecology in various forms (e.g., Liebig's law of the minimum, Shelford's law of tolerance, carrying capacity, Hutchinsonian niche). Huggett (2005) defined ecological thresholds as points or zones at which relatively rapid change occurs from one condition to another. Due to the prevalence of habitat loss and habitat fragmentation throughout the world, researchers have been studying thresholds in landscape structure (e.g., With and Crist 1995, Jansson and Angelstam 1999, Huggett 2005, Denoël and Ficetola 2007).

This dissertation follows the style of Journal of Wildlife Management.

Although the usefulness of identifying thresholds within landscapes for biodiversity and species assemblages is debatable (Lindenmayer and Luck 2005, Denoël and Ficetola 2007), identifying thresholds for target species appears to be practical and useful in conservation and management (Jansson and Angelstam 1999, van der Ree et al. 2003, Radford and Bennett 2004, Denoël and Ficetola 2004, Denoël and Ficetola 2007). The ability to identify thresholds for target species could be useful in conservation and management in multiple ways including setting goals for retention and restoration of the target species' habitat patch size (Huggett 2005).

Most studies searching for ecological thresholds within landscapes have focused on occupancy, which, like density, may not be indicative of productivity (Van Horne 1983). Arnold et al. (1996) reported that golden-cheeked warblers (*Dendroica chrysoparia*) did not occupy patches <10 ha and speculated that 23 ha represented a threshold for consistent production of young. The difference in threshold for occupancy and consistent production of young reveals a potential bias in looking only at occupancy. The danger of this bias could be expressed in conservation and management. For example, in the case of the golden-cheeked warblers, managers may allow a 30-ha patch to be divided into 2 14-ha patches. The remaining patches may allow for occupancy, but may be too small for reproduction. Although there would only be a loss of 2 ha of habitat, there would be a complete loss of production.

Many authors have reported positive correlations between patch size and nest success (Paton 1994, Burke and Nol 2000, Stephens et al. 2003, Rodewald and Vitz 2005). Hypotheses proposed for the positive relationship between patch size and reproductive success includes reduced brood parasitism and predation (Andrén and Angelstam 1988, Wilcove 1985, Robinson et al. 1995) and increased food availability (Burke and Nol 1998, Zanette et al. 2000). Brood parasitism and nest predation are 2 of the leading causes of reproductive failure (Martin 1995). The associations between patch size and brood parasitism and between patch size and predation are uncertain and vary with species and location (Tewksberry et al. 1998). Scientists often attribute such relations to edge effects and the fact that smaller patches of similar shape have higher edge-to-area ratios than larger patches (Andrén and Angelstam 1988, Wilcove 1985). Some researchers found that brood parasitism and nest predation were higher at edges than in interiors of patches (Andrén and Angelstam 1988, Wilcove 1985, Robinson et al. 1995) although others did not (Tewksberry et al. 1998).

Burke and Nol (1998) found correlations between patch size, arthropod densities, and pairing success in ovenbirds (*Seiurus aurocapillus*). Burke and Nol (2000) later concluded that food availability affected reproductive success of ovenbirds. Zanette et al. (2000) observed similar patterns of patch effects on arthropod abundance and reproductive success. They found that arthropod biomass in small patches was about half that of larger patches, females in small patches received 40% less food from mates while on the nests, and females left their nests more often to forage on their own. They also found a shorter breeding season, lower egg weights, and smaller chicks in smaller patches.

The positive relationship between arthropod biomass and patch size may be attributed to edge effects. Microclimate variables influence arthropod abundance (Helle and Muona 1985, Didham et al. 1996). Van Wilgenburg et al. (2001) found that edge-induced changes in microclimate caused a negative response in soil arthropods, but did not affect foliage arthropods.

My objectives were (1) to determine the minimum patch size thresholds of reproductive success for three Neotropical migratory songbirds; (2) to determine whether thresholds for occupancy, territory establishment by males, or pairing success were indicative of thresholds of reproduction; (3) to determine whether the proportion of pairs fledging brown-headed cowbird (*Molothrus ater*) young was related to patch size, and (4) to determine the affects of patch size on food availability (i.e., arthropod abundance). Meeting these objectives could enable natural resource managers to make judicious decisions about where vegetation clearing should be conducted and where to focus future research and conservation efforts.

STUDY SPECIES

I studied the golden-cheeked warbler, black-and-white warbler (*Mniotilta varia*), and white-eyed vireo (*Vireo griseus*). I chose to study these 3 migratory songbirds because they represent varying degrees of sensitivity to forest patch area. Coldren (1998) concluded that golden-cheeked warblers were an area-sensitive species based on observations of positive relationships between reproductive success and patch size and pairing success and patch size. Black-and-white warblers are sensitive to forest area (Hannon 1993, Kricher 1995). Hannon (1993) reported that black-and-white warblers only occupied forest patches >10 ha. White-eyed vireos are not area-sensitive, are habitat generalist, and are known to breed in the interior, exterior, and at the edges of forest stands (Hopp et al. 1995). I included white-eyed vireos, because they are not area-sensitive, which allowed me to observe reproductive activity of an insectivorous migratory species in the full range of patches sizes studied.

All three songbirds are susceptible to cowbird parasitism (Pulich 1976, Hopp et al. 1995, Kricher 1995), are insectivorous (Pulich 1976, Hopp et al. 1995, Kricher 1995), and breed in juniper-oak (*Juniperus-Quercus*) forests (Pulich 1976, personal observation). Golden-cheeked warblers and black-and-white warblers usually rear only a single brood per season, though circumstantial evidence suggests that golden-cheeked warblers occasionally double brood (Ladd and Gass 1999) and black-and-white warblers are suspected of occasionally double brooding (Kricher 1995). Golden-cheeked warblers and black-and-white warblers will attempt to re-nest if their nest is destroyed

(Kircher 1995, Ladd and Gass 1999). Number of broods reared per season by white-eyed vireos is uncertain. White-eyed vireos will also re-nest if nest is destroyed or depredated (person observation). Golden-cheeked warblers require mature stands of juniper-oak (*Juniperus ashei*) forests for breeding, and were listed as endangered in 1990 due to loss of habitat (U.S. Fish and Wildlife Service 1990).

Based on research showing that golden-cheeked warblers and black-and-white warblers are sensitive to patch area and white-eyed vireos are habitat-generalists, I predicted that I would find thresholds of reproductive success for both warblers, but not for white-eyed vireos.

STUDY AREA

I conducted my study on private lands in the Cross Timbers and Prairies and Lampasas Cut Plains plant life areas in east-central Texas (Hatch 2008: 106). The study area consisted of canyons, mesas, and bottomlands composed of alkaline soils and limestone bedrock. The major ecological sites were steep adobe, low stony hill, loamy bottomland, and clay loam (National Cartography and Geospatial Center 2002). Total precipitation between February and May was 43 cm and 61 cm in 2006 and 2007, respectively (National Oceanic and Atmospheric Administration 2006, 2007). Prevalent tree species included Ashe juniper, Texas oak (*Quercus buckleyi*), live oak (*Q. virginiana*), shin oak (*Q. sinuata*), post oak (*Q. stellata*), blackjack oak (*Q. marilandica*), Texas ash (*Fraxinus texensis*), cedar elm (*Ulmus crassifolia*), American elm (*U. americana*), redbud (*Cercis canadensis*), hackberry (*Celtis laevigata*), and pecan (*Carya illinoensis*). About 13% of the study area is composed of patches of mixed juniper-oak forests. The remainder of the area is composed of a mosaic of cropland, rangeland, and developed land. Twenty-five percent of the patches of juniper-oak forest are <3.3 ha, 50% are <6.2, and 75% are <17.7 ha. Patches >17.7 ha make up 81% of the total area of mixed juniper-oak forest. Patches of various sizes are mostly interspersed (Figure 1). Canyons and steep slopes supported most of the mature juniper-oak forests; though, mesa tops and bottomlands contained some forest patches, as well.

In much of east-central Texas, land managers view Ashe juniper (*Juniperus ashei*) as an invasive species (Owens 1996). The removal of juniper surrounding juniper-oak (*Juniperus*-*Quercus*) forests invariably leaves smaller patches.

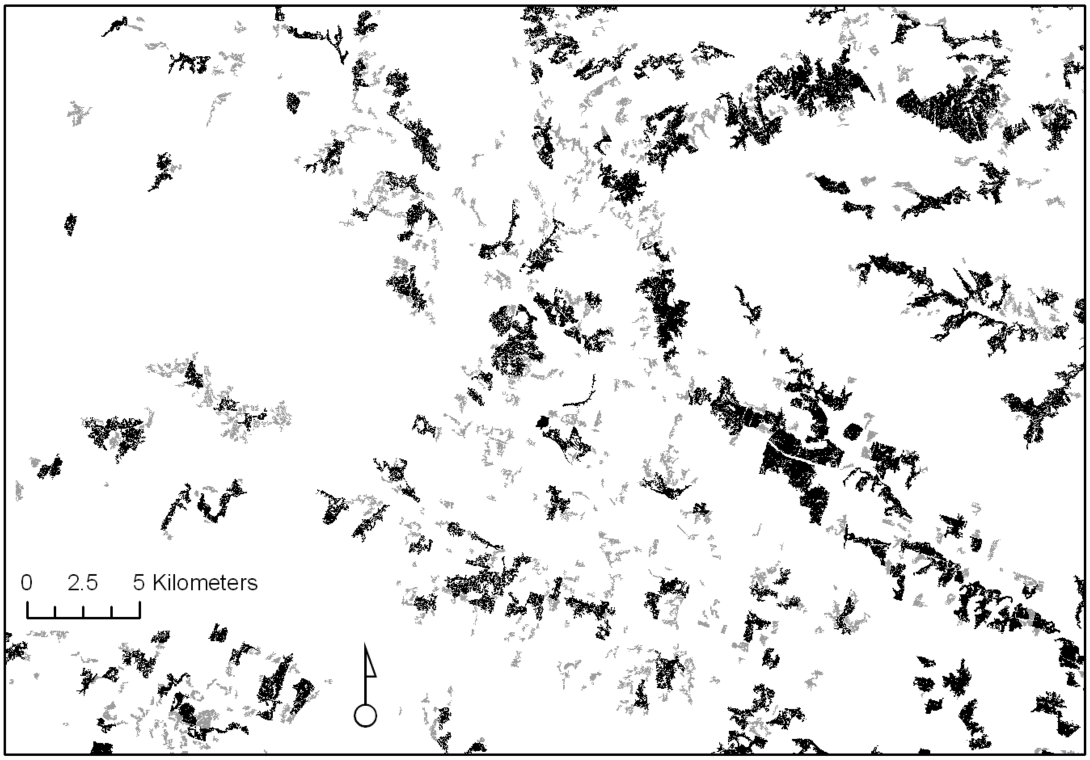

Figure 1. Map showing the interspersion of patch sizes of mature juniper-oak forests. Patches <20 ha are grey and patches >20 are black.

METHODS

Patch size thresholds and reproductive activity

I used a method developed by Vickery et al. (1992), referred to here as the Vickery method, to estimate reproductive activity of birds within patches. The Vickery method allows researchers to predict the reproductive stage of a male or pair based on behavioral observations, observations of host-species fledglings, and observations of cowbird fledglings without the time constraints of locating and monitoring nests (Vickery et al. 1992, Christoferson and Morrison 2001). Using the method proposed by Vickery et al. (1992), Christoferson and Morrison (2001) correctly predicted the outcome of 80 to 92% of nests for 3 songbirds.

The Vickery method includes assigning ranks to males or pairs that represent the most advanced stage of reproductive activity reached during the season. I used a modified version of Vickery et al's (1992) ranking system to meet my objectives. Ranks included occupancy (rank 1; Table 1), territory formation (rank 2; Table 1), pair formation and fledging cowbird young (rank 3; Table 1), and fledgling host-species young (rank 4; Table 1). I identified territorial males and pairs within the patches using a 3-step process. First, to cover all patches quickly and thoroughly, I systematically placed transects throughout patches and walked each transect twice, recording locations of each individual of the target species. I placed transects ~100 m apart in the patches so that no area in the patch was >100 m from a transect. Second, beginning on 1 April, I used spot-mapping to delineate territories. Based on preliminary data collection in the

Table 1. Description of ranks of reproductive activity assigned to each golden-cheeked warbler, black-and-white warbler, and white-eyed vireo. Ranks were based on individuals' behaviors and other evidence of breeding activity and modified from Vickery et al. (1992).

Rank	Description
1	Present
2	Male present >4 weeks (considered territorial)
3	Male and female present >4 weeks (considered paired)
	Evidence of nest building, male carrying food to presumed female on nest, or the female laying or incubating eggs
	Female carrying food to presumed nestlings
	Cowbird parasitized nest
4	Host-species fledgling with pair (considered successful)

study region (J. A. Butcher, Texas A&M University, unpublished data) most males establish territories and formed pairs by 1 April. I recorded an average of 41 points per territory over an average of 12 visits per territory (i.e., ~3 registration points per territory per visit), which exceeded recommendations by International Bird Census Committee (1970). I recorded the distance and direction to other individuals and locations where intraspecific interactions occurred. I entered all locations and interactions into a geographical information system (GIS) to delineate territory boundaries for males and pairs. Because I attributed reproductive success to the entire patch, exact delineation of breeding territories was not imperative to the study. That is, if, within a patch, ≥1 pair successfully fledged ≥1 host-species young I considered the patch to be above the threshold of patch size in which reproduction would likely occur. Third, during the third week in April, around the approximate date chicks begin to fledge in the region (J. A. Butcher, unpublished data, Kricher 1995, Ladd and Gass 1999) I conducted searches for fledglings within each territory. I searched each territory for fledglings an average of 12 times. I spent an average of 32 (SE = 0.5) min within each territory each time I searched for fledglings. During all 3 steps I recorded GPS points on males, females, and fledglings and recorded observations for the Vickery method.

I defined a patch as a stand of juniper-oak forest that was at least 8 m from other such stands (Rich et al. 1994, Horne 2000). Rich et al. (1994) found that corridors as narrow as 8 m contributed to negative edge effects on local breeding songbirds by attracting avian nest predators and cowbirds. Horne (2000) found that openings as narrow as 10 to 20 m might result in the loss of breeding habitat for golden-cheeked warblers. Patches met criteria for vegetation characteristics set forth by the Texas Parks and Wildlife (2005) for suitable habitat for golden-cheeked warblers. I included the criteria that Texas oak must be present in every patch, because of its importance as foraging substrate for golden-cheeked warbler (Kroll 1980), black-and-white warbler, and white-eyed vireo in my study area (personal observation). I chose to base the definition of a patch on habitat of golden-cheeked warblers because of their endangered status. Black-and-white warblers and white-eyed vireos, however, are known to breed within patches of habitat suitable for golden-cheeked warblers (personal observation). The definition of a patch for white-eyed vireos would likely encompass a greater area because of their ability to breed outside of the area included in the definition of a patch for golden-cheeked warblers. The reason for including them in the study is because they are not sensitive to patch area as defined for golden-cheeked warblers and black-and-white warblers.

I searched for thresholds in patches that ranged from 2.9 to 27.7 ha. The minimum size selected represented the approximate mean territory size of golden-cheeked warblers (Pulich 1976, Weinberg et al. 1996). I based the maximum size on knowledge that golden-cheeked warblers are successful in patches of >23 ha (Arnold et al. 1996). I exceeded the 23 ha patch size reported by Arnold et al. to ensure that the range contained the threshold of reproductive success. All available patches of mature juniper-oak forests that existed entirely on accessible private property in Bosque, Coryell, and Hamilton counties made up the sampling frame. I used 2004 digital orthoquads (DOQQ) and a geographical information system (GIS) to delineated and calculated area of potential patches. I drew polygons around forest patches that were visible on the DOQQs and then visited the patches to ensure that they met the above criteria.

I recorded locations, sex, age (adult or fledgling), and Vickery rank values of all individuals of the target species in 12 patches over 2 seasons. I observed 5 patches from 15 March to 6 July 2006 and 7 patches from 16 March to 22 June 2007. I visited each patch an average of 24 (SE = 2.6) days during each season. I attempted to visit each breeding territory every 3 days. The 12 patches represent a census of accessible patches in the 4-county study area.

Arthropod biomass

I collected branch clippings from 22 April to 10 May 2006 and from 24 April to 7 May 2007 to assess the relationship between patch size and arthropod abundance (Cooper and Whitmore 1990, Keane and Morrison 1999). Johnson (2000) found that branch clipping is an effective way of sampling food availability for foliage-gleaning species. Branch clipping included quickly placing a plastic bag over a branch, clipping the branch from the tree, freezing the sample to kill the arthropods, drying the samples at 60 ºC for ~6 days, separating the arthropods from the leaves, and weighing both the leaves and the arthropods (Johnson 2000, Rodewald and Vitz 2005). I randomly placed sampling stations by overlaying each patch with a 100 × 100 m-cell grid that had a random origin, assigning each intersection a number, and using a random number table to select stations. The number of sampling stations was proportional to patch size, although I took more samples per patch during 2007 than in 2006.

I used 3 techniques to minimize variability in arthropod samples. First, I collected branch samples during the period of the breeding season when most golden-cheeked warblers, black-and-white warblers, and white-eyed vireos have nestlings and fledglings in the study region (personal observation). Limited food availability during the nestling and fledgling stages can negatively affect survival of young (Simons and Martin 1990). Wharton et al. (1996) found that arthropods in juniper-oak forests increased in abundance from March through the end of April, followed by a small decrease in May, and then remained stable through June. Second, I set the height above ground where I took samples to 2 m. Collecting samples at 2 m allowed for consistency in placing the bag over the branch so that few arthropods would escape. Sampling at 2 m was also justifiable based on arthropod distribution and feeding behavior of the birds. Wharton et al. (1996) found that the arthropod species that they collected in large numbers in juniper-oak forests tended to show no preference for any particular height on the tree. Beardmore (1994) reported male golden-cheeked warblers spent more time foraging below 3 m while female golden-cheeked warblers spent more time foraging above 5 m. I combined Beardmore's (1994) categories of male and female foraging times and found that male and female golden-cheeked warblers spend 27% of the time foraging below 3 m, 28% foraging between 3 and 5 m, and 45% foraging above 5 m.

Third, I collected samples from 2 tree species. I chose to collect from Ashe juniper and Texas oak trees, because Wharton et al. (1996) found that most species of arthropods showed preferences for particular tree species. Ashe juniper and Texas oak comprise 2 of the 4 species that Beardmore (1994) reported that golden-cheeked warblers foraged in the most, the other 2 tree species were live oak and cedar elm, neither of which are as abundant as Ashe juniper and Texas oak in my study area (Juarez Berrios 2005). Wharton et al. (1996) stated that several insect species in all of the major orders of arthropods preferred oak, specifically Texas oak. Based on preliminary work in the study region, black-and-white warblers and white-eyed vireos forage at 2 m and in both Ashe juniper and Texas oak (personal observation). All 3 songbird species feed on a large array of arthropod species and forage to some extent on leaves and branches (Morse 1970, Pulich 1976, Hopp et al. 1995).

Data analysis

The criteria described above and the fact that my work was on private land limited my sample size. I had too few samples (patches) to analyze the reproductive activity with multinomial logistic regression; therefore, I used descriptive statistics and presented data in tables and figures.

I tabulated the relationship between patch size and the 4 ranks of reproductive activity for each target species. The 4 ranks of reproductive activity included (1) occupancy, (2) territory establishment, (3) pairing success, and (4) reproductive success. To determine whether occupancy, territory establishment, or pairing success could be used as an indicator of reproductive success I compared the thresholds of each rank to the threshold of reproductive success. I determined that the threshold of a given rank of reproductive activity (e.g., occupancy) was a good indicator of the threshold of reproductive success if the minimum patch size threshold of a given activity (e.g., occupancy) was equal to the minimum patch size threshold for reproductive success.

I calculated the mean and corresponding 95% confidence intervals for biomass of arthropods >1 mm in length for each tree species in each patch. To determine whether there was evidence of an edge effect I analyzed the relationship between arthropod biomass and distance from edge by presenting a scatterplot and calculating Pearson's correlation coefficients. To determine whether patch size influenced arthropod biomass I analyzed the relationship between patch size and arthropod biomass by calculating Pearson's correlation coefficients. To compare food availability in patches where success occurred to food availability in patches where success did not occur, I calculated mean and standard error of arthropod biomass in patches above and below the thresholds of reproductive success for each species. I was unable to run further statistics because the number of patches where success occurred was too small for the songbirds exhibiting thresholds of reproductive success.

RESULTS

Thresholds of reproductive success

During the 2 seasons I delineated 24 golden-cheeked warbler territories, 9 black-and-white warbler territories, and 47 white-eyed vireo territories. I observed golden-cheeked warblers in 11 of 12 (92%) patches including the smallest patch studied (Table 2). Male golden-cheeked warblers established territories in all 11 patches in which they occupied and established pairs in 7 (64%) of the patches where they established territories. Pairs fledged ≥1 young only in patches >15 ha, and no more than 1 pair formed in any patch ≤15 ha. In patches >15 ha 15 of 17 (88%) males were paired and 13 of 15 pairs (86%) fledged ≥1 young. In patches ≤15 ha 3 of 7 (42%) males were paired. Despite the presence of brown-headed cowbirds in the patches, I observed no evidence of cowbird parasitism on golden-cheeked warblers.

I observed black-and-white warblers in 7 of 12 (58%) patches including the smallest patch studied (Table 2). Males established territories in 2 patches; both patches were >15 ha. Seven of 9 (78%) territorial males paired and all pairs fledged ≥1 young. A cowbird parasitized one pair; however, the pair fledged their own young as well. The incident of cowbird parasitism occurred in the largest patch studied.

I observed white-eyed vireos in 11 of 12 (91%) patches including the smallest patch studied (Table 2). Males established territories in all patches in which I observed them. Ten of 12 (83%) patches contained pairs. Pairs fledged ≥1 young in patches >4.1 ha. In patches >4.1 ha

Table 2. Occupancy (Y = yes) and number of golden-cheeked warblers, black-and-white warblers and white-eyed vireos males that reached each level of reproductive activity based on measurements using the Vickery method (Vickery et al. 1992). I made observations in 12 patches ranging from 2.9 to 27.7 ha. Threshold of reproductive success observed for golden-cheeked warbler (dashed line) and black-and-white warbler (dot-dashed line).

	Golden-cheeked warbler				Black-and-white warbler				White-eyed vireo			
Patch size (ha)	Occupancy[1]	Territorial[2]	Paired[3]	Successful[4]	Occupancy[1]	Territorial[2]	Paired[3]	Successful[4]	Occupancy[1]	Territorial[2]	Paired[3]	Successful[4]
27.7	Y	6	5	5	Y	5	5	5	Y	9	9	6
22.2	Y	3	3	3	Y				Y	6	6	4
21.1	Y	4	3	2	Y				Y	10	9	6
20.1	Y	4	4	3	Y	4	2	2	Y	7	5	3
15.0									Y	2	1	
11.9	Y	1							Y	4	4	4
10.8	Y	1	1		Y				Y	2	1	
8.9	Y	1							Y	3	3	2
4.4	Y	1	1						Y	2	2	2
4.1	Y	1	1		Y							
3.2	Y	1							Y	1	1	
2.9	Y	1			Y				Y	1		

[1] Observed individual in the patch during the breeding season

[2] Number of males that established and defended a territory for >4 weeks

[3] Number of males observed with a female for >4 weeks

[4] Number of pairs that successfully fledged ≥1 offspring

40 of 45 (88%) males were paired and 27 of 40 (68%) pairs fledged ≥1 young. In patches ≤4.1ha only 1 of 2 territorial males paired. Two white-eyed vireo pairs fledged cowbird young. One pair was in the 2.9-ha patch and the other was in the 15-ha patch. The number of pairs that fledged young was linearly related to patch size (Figure 2).

Golden-cheeked warblers and black-and-white warblers were not detected in the 15.0 ha patch (Table 2). The reason is uncertain, but was not likely caused by a landscape configuration. The 15.0 ha patch was within 10 m of 2 neighboring patches, and mixed juniper-oak forest made up 13% of the area that fell within a 400 m buffer

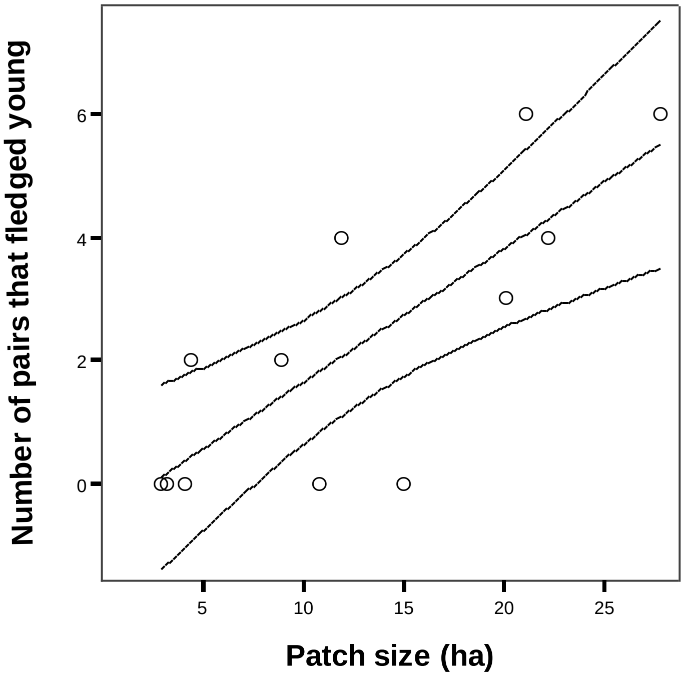

Figure 2. The number of white-eyed vireo pairs that fledged young was linearly related to patch size ($r^2 = 0.63$). Lines show mean and 95% prediction interval.

around the patch. The average area of juniper-oak forest surrounding patches was 24.3 ha (n = 12, SE = 3.4). Two patches with less juniper-oak forest surrounding the patch were occupied by golden-cheeked warblers (4.4 and 21.1 ha patches) and 1 patch was occupied by the black-and-white warbler (21.1 ha patch).

Arthropod biomass

I collected 209 branch clippings each from juniper and oak trees (Table 3). Arthropod biomass in Ashe juniper and Texas oak did not correlate with patch size (r = 0.13, r = 0.12, respectively; Fig. 3 *a, b*) or distance from edge (r = 0.03, r = 0.11, respectively; Fig. 4 *a, b*). The mean arthropod biomass above the observed minimum patch size threshold of reproductive success for golden-cheeked warblers and black-and-white warblers was 0.16 mg/g (n = 4, SE = 0.04) and 0.50 mg/g (n = 4, SE = 0.26) in juniper and oak trees, respectively. Below the observed minimum patch size threshold of reproductive success the mean arthropod biomass was 0.14 mg/g of leaves (n = 8, SE = 0.03) and 0.45 mg/g of leaves (n = 8, SE = 0.15) in juniper and oak trees, respectively.

Table 3. Arthropod biomass (mg/g of leaves) collected from branch clippings taken in 12 patches of juniper-oak (*Juniperus-Quercus*) forest in east-central Texas. I collected branch clippings 2 m above ground from Ashe juniper (*Juniperus ashei*) and Texas oak (*Quercus buckleyi*) from 22 April 2006 to 10 May 2006 and from 24 April 2007 to 7 May 2007 to assess the relationship between patch size and arthropod biomass.

	Ashe juniper		Texas oak	
Patch size (ha)	N	Mean (SE)	N	Mean (SE)
27.7	31	0.21 (0.05)	30	0.41 (0.12)
22.2	21	0.18 (0.04)	21	2.39 (0.43)
21.1	29	0.05 (0.01)	27	0.41 (0.11)
20.1	25	0.19 (0.06)	25	0.21 (0.08)
15.0	21	0.09 (0.03)	21	0.36 (0.13)
11.9	18	0.31 (0.12)	17	2.42 (0.43)
10.8	15	0.09 (0.02)	16	0.42 (0.30)
8.9	14	0.11 (0.05)	14	0.55 (0.33)
4.4	9	0.05 (0.01)	9	0.57 (0.37)
4.1	10	0.18 (0.09)	10	1.42 (0.48)
3.2	10	0.15 (0.07)	10	0.33 (0.19)
2.9	9	0.17 (0.09)	9	0.06 (0.05)

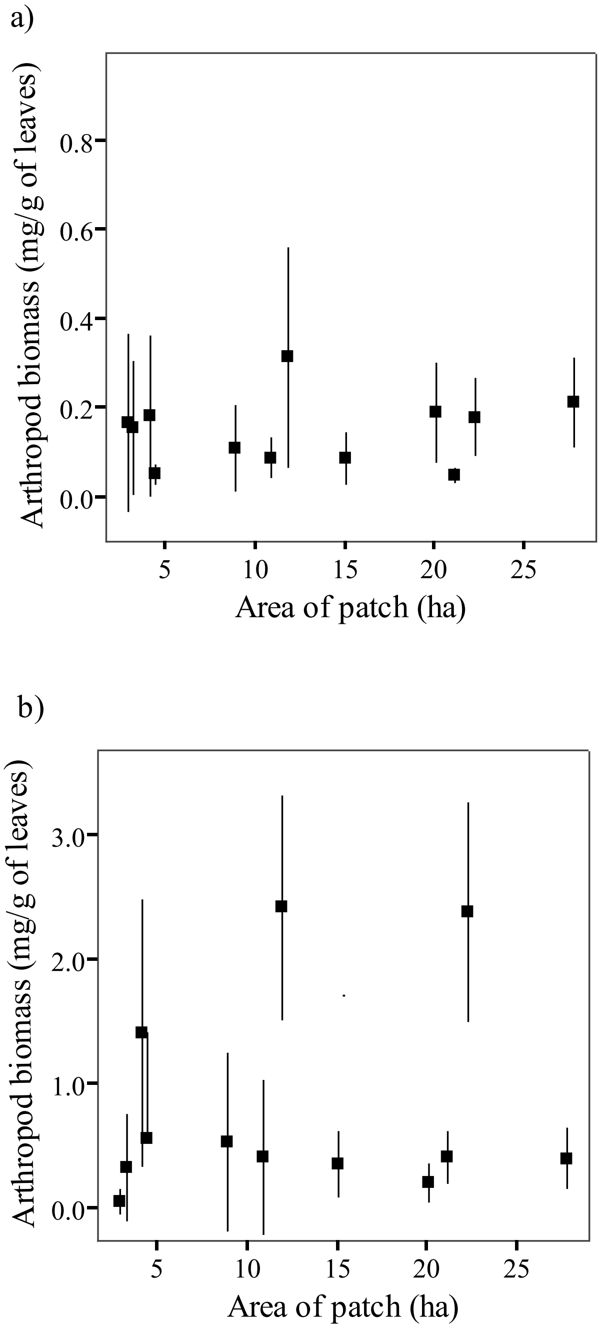

Figure 3. Arthropod biomass in each patch obtained from branch clippings of Ashe juniper (a) and Texas oak (b) during the average nestling and fledgling stages of golden-cheeked warblers, black-and-white warblers, and white-eyed vireos in east-central Texas. Squares represent mean arthropod biomass and lines represent 95% confidence intervals. Notice that the ordinates are of different scales due to the greater arthropod biomass on Texas oak than on Ashe juniper.

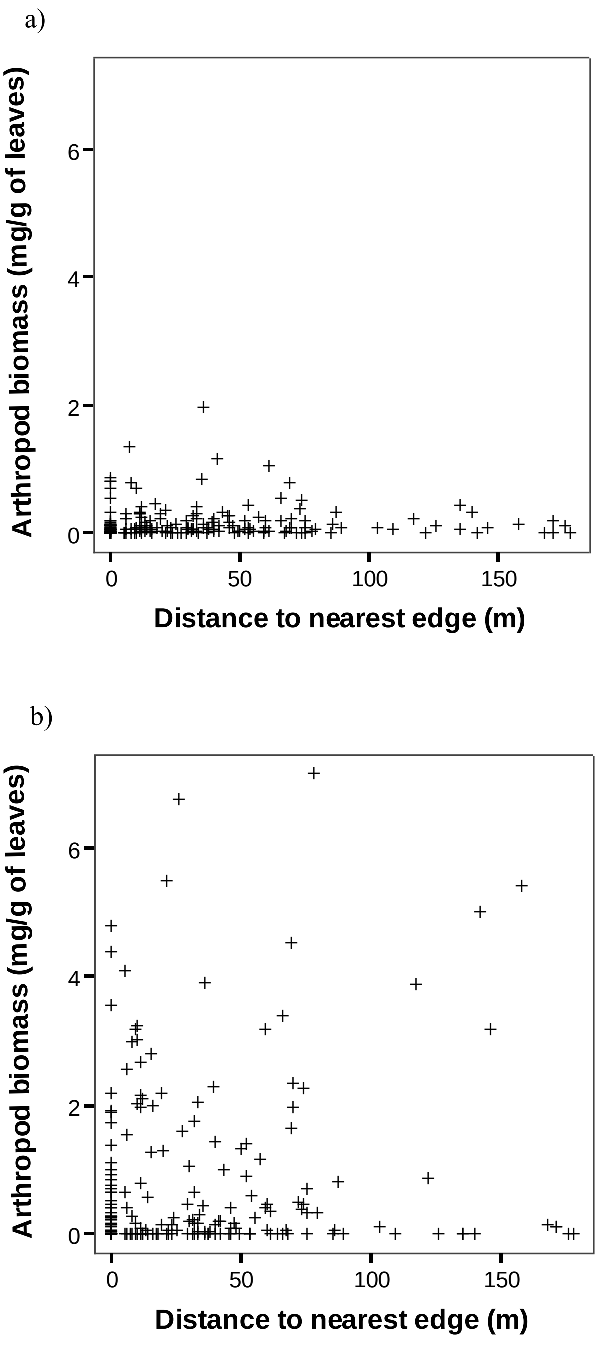

Figure 4. Arthropod biomass relative to distance from edge obtained from branch clippings of Ashe juniper (a) and Texas oak (b) during the average nestling and fledgling stages of golden-cheeked warblers, black-and-white warblers, and white-eyed vireos in east-central Texas.

DISCUSSION AND CONCLUSIONS

I found minimum patch size thresholds of reproductive success for golden-cheeked warblers and black-and-white warblers. Minimum patch size thresholds of occupancy, territory establishment, and pairing were not indicative of minimum patch size thresholds of reproduction. The observed minimum patch size threshold of reproductive success for golden-cheeked warblers and black-and-white warblers was between 15 and 20.1 ha. In a study conducted south of my study area, Arnold et al. (1996) observed a similar trend for golden-cheeked warblers. They reported that golden-cheeked warblers reliably produced young in patches >23 ha in size. To my knowledge no study has been published relating productivity of black-and-white warblers to patch size. Hannon (1993), however, found that black-and-white warblers were sensitive to patch size; she detected black-and-white warblers only in patches >10 ha.

As expected, I did not find a minimum patch size threshold of reproductive success for white-eyed vireos. The fact that there was no success in the 3 smallest patches can be explained by natural variation regardless of patch size. Territory establishment and pairing success increased linearly with patch size. Bender et al. (1998) predicted that population changes in generalist species that use both edges and interiors of forest patches would be accounted for by habitat loss alone, regardless of fragmentation.

Although research shows that birds nesting in smaller patches experience higher predation and parasitism (Hoover et al. 1995), neither appeared to be a proximate cause for the thresholds I observed. Parasitism within the patches was low and was not related to patch size in the range of patches that I studied. Low parasitism frequency was surprising because of the ubiquity of brown-headed cowbirds in the study area. Cowbirds were observed at 88% of survey stations within the study area (Juarez Berrios 2005), and >80% of black-capped vireo and white-eyed vireo nests in shrubs and trees surrounding my study patches were parasitized by brown-headed cowbirds (S. L. Farrell, Texas A&M University, unpublished data). Although the relationship between forest cover and parasitism is highly variable (see Tewksberry et al. 2006), the high magnitude of difference between parasitism inside and outside of the juniper-oak forest patches in my study area suggest a relationship that should be studied.

Although I did not study predation, I would expect that more white-eyed vireos would have failed in the patches below the thresholds of the warblers if predator activity caused the observed thresholds. Ultimately, adaptations by the warblers could have caused them to select larger patches because of the influence of predation in small patches in the historic past. Fontaine and Martin (2006) found that some migratory songbirds have the ability to assess predator activity and adjust breeding location accordingly.

High success above the threshold suggests that predation and parasitism were not limiting factors within larger patches. Success of golden-cheeked warbler pairs above the threshold (86% of pairs fledged young) was similar to populations at Fort Hood, Texas, (87.8% of pairs fledged young; Anders and Marshall 2005) where intensive cowbird control has been in effect since 1991 (Eckrich et al. 1999). I observed 100% of black-and-white warbler pairs fledged young.

I found no relationship between patch size and arthropod biomass. Nour et al. (1998) found that neither evidence of caterpillars (frass fall) nor provisioning rates for young great tits (*Parus major*) and blue tits (*P. caeruleus*) were correlated with patch size. Similarly, Buehler et al. (2002) found that food availability and provisioning rates of hooded warblers (*Wilsonia citrina*) did not relate to patch size. My findings, along with Nour et al. (1998) and Buehler et al. (2002), are contradictory to Burke and Nol (1998, 2000) and Zanette et al. (2000) who found that food availability was correlated with patch size and ultimately reproductive success. The differences can be explained by sampling objectives and thus sampling protocols. The 3 focal species of my study along with hooded warblers studied by Buehler et al. and the 2 species of tits studied by Nour et al. all feed above ground, therefore arthropod sampling was conducted above ground. Burke and Nol (1998) studied ovenbirds (*Seiurus aurocapillus*) and Zanette et al. (2000) studied eastern yellow robins (*Eopsaltria australis*) both of which are ground-foragers and, thus, sampling occurred at the ground level. Van Wilgenburg et al. (2001) reported that soil-dwelling arthropods responded negatively to edge effects whereas canopy-dwelling arthropods in the same forests showed no response to edge.

I did not differentiate patch size affects on specific groups of arthropods. It is possible that particular arthropod families, orders, or species are important to each bird species, and that patch size affected those arthropod groups. Although Lepidoptera larvae comprised the greatest percentage of any one order of arthropods in stomach contents for all 3 songbird species, the contents, along with observational studies, revealed that all 3 species feed on a large variety of arthropods (Nolan and Wooldridge 1962, Pulich 1976, Kricher 1995).

Because there was no relationship between patch size and arthropod biomass, there was little reason to believe that food availability was the cause of the observed thresholds. The patterns related to the thresholds may offer some insight to possible causes for the thresholds of reproductive success. Besides fledging young, an obvious difference between reproductive activity above and below the threshold of reproductive success was number of territories established and number of pairs formed; for all 3 species success occurred only in patches where >1 pair was formed. Researchers have noticed that some territorial species aggregate their territories even when surrounding unoccupied habitat exists (Svärdson 1949, Hildén 1965, Stamps 1988). Two theoretical reasons for aggregation of territories are (1) conspecifics may act as cues for settling individuals and (2) there might be some type of benefit in living within an aggregation (e.g., predator protection, access to mates; Muller et al. 1997). My results showed what appeared to be clumping for golden-cheeked warblers and black-and-white warblers. The aggregation of black-and-white warblers into 2 patches despite occupying multiple patches suggests conspecific attraction, defined as aggregation of territories caused by apparent attraction to neighbors (Stamps 1988, Ahlering and Faaborg 2006, Campomizzi et al. 2008). Campomizzi et al. (2008) reported evidence of conspecific attraction in golden-cheeked warblers in the same study area. Without experiments controlling for habitat quality, predator activity, and other variables that influence habitat selection, conspecific attraction as a means of habitat selection is difficult to discern from other habitat cues (e.g., predator activity, food availability; Stamps 1988).

Although determining success by searching territories may express season-long productivity more accurately than nest monitoring (Anders and Marshall 2005), fledging young does not necessarily culminate in higher fitness (number of offspring that successfully reproduce). I relocated fledglings throughout the season, but at the end of the season I was unable to determine whether juveniles that fledged in the patches were depredated, starved, or survived to migrate south. Further research on survival of juveniles, recruitment, and offspring breeding success could help address relationships between patch size and fitness.

To date, most researchers used occupancy of the target species as the response variable for thresholds. My research showed that minimum patch size thresholds for occupancy were not indicative of the thresholds of reproductive success, and minimum patch size thresholds for territory establishment and pair formation were not indicative of patch size thresholds for reproductive success. Management and conservation practices for target species based on thresholds of occupancy, territory establishment, or pair formation could be dangerous. Some may argue, and rightly so, that preserving patches based on occupancy, territory establishment, or pair formation will include patches where reproduction could occur. However, the danger comes not when managers attempt to conserve all patches above the threshold of occupancy or pair formation (that would be ideal), but when managers use the minimum patch size threshold of occupancy as a basis for allowing larger patches to be fragmented into patches that may fall below the patch size threshold of reproductive success.

Thresholds are an integral part of ecological theory (e.g., Shelford's law of tolerance, Hutchinsonian niche). Studies show that thresholds of habitat distribution, amount, and configuration of a species' habitat in a landscape influence population dynamics (Fahrig and Merriam 1994, Morrison et al. 1998: 48–49, Hokit and Branch 2003, Denoël and Ficetola 2007). The ability to identify thresholds of reproductive success for target species could be useful in conservation and management in multiple ways including setting goals for retention and restoration of target species' habitat patch size (Huggett 2005).

Management implications

Because both warblers have relatively uniform habitat requirements across their distribution ranges (Kricher 1995, Ladd and Gass 1999), patch size relationships observed in this study should hold across much of their ranges. Managers involved in juniper clearing in east-central Texas particularly, and in forest removal in general, should be cautious not to decrease patches below 20 ha. Because patches below the threshold of reproductive success were occupied by golden-cheeked warblers, research is needed to determine the role that such patches play in population dynamics of golden-cheeked warblers.

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Appendix G

Red imported fire ants can decrease songbird nest survival

SHORT COMMUNICATIONS

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RED IMPORTED FIRE ANTS CAN DECREASE SONGBIRD NEST SURVIVAL

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Abstract. Invasive species are often implicated in population declines of native species because of predation. The red imported fire ant (*Solenopsis invicta*) has been documented to prey on songbird nests. We conducted a replicated manipulative experiment to determine the decrease in nest survival caused by *S. invicta*. In 2006 and 2007 we monitored 71 nests, 44 of the White-eyed Vireo (*Vireo griseus*) and 27 of the Black-capped Vireo (*V. atricapilla*), in nine patches of 36–103 ha each in central Texas. We prevented *S. invicta* from preying on nests by applying insect-specific chemical and physical barriers at individual nests. Excluding *S. invicta* increased nest survival from 10% to 31% for the White-eyed Vireo and from 7% to 13% for the Black-capped Vireo. Our results suggest the decrease in nest survival of songbirds susceptible to predation by *S. invicta* may be substantial in the areas this ant occupies.

Key words: Black-capped Vireo, invasive species, nest predator, nest success, Solenopsis invicta, *Texas,* Vireo atricapilla, Vireo griseus, *White-eyed Vireo.*

La Hormiga Importada *Solenopsis invicta* Puede Causar Disminuciones en la Supervivencia de los Nidos de Aves Canoras

Resumen. Las especies invasoras frecuentemente están implicadas en las disminuciones de las poblaciones de especies nativas debido a la depredación. Se ha documentado que la hormiga importada *Solenopsis invicta* depreda nidos de aves canoras. Realizamos un experimento de manipulación con réplicas para determinar la disminución en la supervivencia de los nidos causada por *S. invicta*. En 2006 y 2007, monitoreamos 71 nidos (44 de *Vireo griseus* y 27 de *V. atricapilla*) en nueve parches de 36–103 ha en el centro de Texas. Impedimos que las hormigas depredaran los nidos aplicando barreras físicas y químicas específicas para insectos. La exclusión de *S. invicta* aumentó la supervivencia de los nidos del 10% al 31% en *V. griseus* y del 7% al 13% en *V. atricapilla.* Nuestros resultados sugieren que la disminución en la supervivencia de los nidos de aves canoras que son susceptibles a la depredación por parte de *S. invicta* podría ser considerable en las áreas que ocupa esta hormiga.

Invasive species are often implicated in population declines of native species because of competition and predation (Caughley and Gunn 1996). Invasive ant species are more abundant and forage more intensely than native ants (Holway et al. 2002). The red imported fire ant (*Solenopsis invicta*) is an invasive ant considered a threat to native species in the United States (Taber 2000, Allen et al. 2004), Caribbean (Davis et al. 2001), Australia (Moloney and Vanderwoude 2002, 2003), New Zealand (Pascoe 2001), and China (Zhang et al. 2005).

In the United States researchers have documented the detrimental effects of *S. invicta* on native arthropods (Porter and Savignano 1990, Gotelli and Arnett 2000), birds, mammals, and herpetofauna (Allen et al. 2004). *Solenopsis invicta* is a known predator of nests of several songbird species in the Mississippi alluvial valley (Twedt et al. 2001) and of the Black-capped Vireo (*Vireo atricapilla*; Stake and Cimprich 2003) and Barn Swallow (*Hirundo rustica*; Kopachena et al. 2000) in Texas. Removal of *S. invicta* from areas where ground-nesting birds breed increased nestling or fledgling survival of colonial water birds (Drees 1994), the Least Tern (*Sternula antillarum*; Lockley 1995), and Northern Bobwhite (*Colinus virginianus*; Allen et al. 1995, Mueller et al. 1999). To our knowledge no one has yet evaluated the effects of *S. invicta* on nest survival of birds nesting off the ground, particularly that of songbirds.

The potential negative effects of *S. invicta* on nest survival could be important to productivity because nest predation is the primary cause of nest failure for many songbird species (Nice 1957, Ricklefs 1969, Martin 1993). Removing nest predators to improve nest survival is occasionally suggested as a management technique to aid in recovering threatened or endangered songbirds (U. S. Fish and Wildlife Service 1991, Cain et al. 2003). Predator–prey interactions involving songbird nests, however, are complex; some predator removal experiments have led to increased nest survival (Jackson 2001, Schmidt et al. 2001), whereas others have resulted in little change (Beauchamp et al. 1996, Schmidt et al. 2001).

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The Condor, Vol. 111, Number 3, pages 534–537. ISSN 0010-5422, electronic ISSN 1938-5422. © 2009 by The Cooper Ornithological Society. All rights reserved. Please direct all requests for permission to photocopy or reproduce article content through the University of California Press's Rights and Permissions website, http://www.ucpressjournals.com/reprintInfo.asp. DOI: 10.1525/cond.2009.090066

We investigated the effect of *S. invicta* on nest survival of the federally endangered Black-capped Vireo and the co-occurring White-eyed Vireo (*V. griseus*). We selected these species because *S. invicta* has been identified as a primary nest predator of the Black-capped Vireo in central Texas (Stake and Cimprich 2003) and the White-eyed Vireo nests in the same habitat. We conducted a replicated manipulative experiment to determine if *S. invicta* causes nest survival over the entire nesting cycle to decrease by 10% or more. We tested for a reduction in nest survival of ≥10% because sensitivity analyses of other songbirds suggest a reduction of this size may be biologically significant for populations (Powell et al. 1999, Donovan and Thompson 2001).

METHODS

STUDY AREA

We studied both vireo species on nine sampling units located on 11 private properties in the Leon River watershed in Coryell County in east-central Texas. Because we had permission to access only some private properties, sampling-unit selection was limited. The nine sampling units, 36–103 ha each, were the only patches of vegetation where Black-capped Vireos had been detected on previous presence–absence surveys of 33 properties within a 140 000-ha area. Vegetation on sampling units consisted of mid-successional woody vegetation similar to that of areas occupied by Black-capped Vireos in the nearby Lampasas Cut Plains (Grzybowski et al. 1994, Bailey and Thompson 2007) and ecotones between mature oak–juniper (*Quercus–Juniperus*) woodland and grassland. In the study area, private lands were used primarily for ranching, farming, and hunting.

Our target population consisted of the Black-capped Vireos nesting in the study area. We included the White-eyed Vireo, which occurs with the Black-capped Vireo in this area, to broaden the information on effects of *S. invicta* on nesting songbirds and to increase sample size. We assumed *S. invicta* would prey opportunistically on any songbird nest it finds while foraging (Wilson 1962) because all songbird nestlings provide a similar source of protein (Stein et al. 1990). We included nests found 0.4 to 4.2 m above the ground because that is the height range of Black-capped Vireo nests in our study area and typical of the placement of Black-capped Vireo nests in general (Graber 1961, Grzybowski 1995).

MANIPULATIVE EXPERIMENT

We used behavioral cues and systematic searching (Martin and Geupel 1993) to locate active songbird nests from 15 March to 31 July in 2006 and 2007. We visited active nests every 3 or 4 days to determine outcome (i.e., nest fledged at least one young or failed). We considered a nest successful if we saw adults carrying food to fledglings or if we detected fledglings by sight or sound near nests that on our previous visit had large nestlings near fledging age. We considered nests failed if they were empty after having eggs or if we did not detect fledglings or observe adults carrying food near nests that previously had nestlings.

We assigned each nest to either the treatment or control group by using a probabilistic starting point (coin toss) for the first nest found, followed by systematic assignment (alternating between treatment and control) for each additional nest. This method of assigning treatments allowed treatment and control nests to be interspersed in space and time within each sampling unit (sensu Hurlbert 1984). We did not include nests in our analysis if they failed before treatment could be applied or if we could not determine whether young had fledged from the nest. We treated individual nests rather than reducing numbers of *S. invicta* by broadcasting ant-specific poison baits, as is typically done in research on *S. invicta* (e.g., Martin et al. 1998, Calixto et al. 2007). Our method avoided the unknown effects of removing *S. invicta* from the nest–predator assemblage (e.g., cause a population response of other nest predators).

We prevented *S. invicta* from preying on treatment nests by applying a chemical barrier (Spiral Wrap Arinix Nix of America, San Jose, CA) and a physical barrier (Tree Tanglefoot Pest Barrier, Tanglefoot Co., Grand Rapids, MI) to branches supporting nests. The Spiral Wrap is a 5-cm-long flexible plastic containing 8.6% permethrin, a pesticide of low toxicity, and was designed to prevent small insects from entering automobile vents. The Spiral Wrap releases permethrin slowly from the plastic and stops ants from crossing. Tanglefoot is a gum resin that catches crawling insects, stopping them from climbing trees and damaging agricultural products. We applied the Tanglefoot Pest Barrier to the branch ≥0.25 m from each treatment nest. We then placed a Spiral Wrap on top of the Tanglefoot Pest Barrier, allowing the Tanglefoot to fill gaps between the branch and Spiral Wrap. Vireo nests are built on a single isolated branch, enabling the ant barrier to block the only access point from which *S. invicta* could reach the nest. We assumed the barrier did not deter other predators (e.g., snakes, birds, mammals) from preying on nests because these predators could easily maneuver past the insect-specific 5-cm barrier. If the barrier deterred other nest predators, then observed treatment effects would be due to treatment nests being protected from *S. invicta* plus other nest predators. To control for potential effects of applying the Spiral Wrap at treatment nests, such as attracting nest predators, we attached an inert Spiral Wrap to branches supporting control nests.

STATISTICAL ANALYSES

We defined nest survival as the probability that a nest fledges at least one young and daily survival rate as the probability that a nest survives one day. We determined the magnitude of nest predation by *S. invicta* from estimates calculated with the nest-survival analysis (Dinsmore et al. 2002) in the program MARK (MARK 5.1, G. C. White, Colorado State University, Fort Collins, CO). We estimated daily survival rate for each species of vireo and each treatment separately. We estimated nest survival over the entire nesting cycle by raising each estimate of daily survival rate to the power of the number of days in the nesting period (Mayfield 1961), 27 days for the White-eyed Vireo (Hopp et al. 1995), 30 days for the Black-capped Vireo (Grzybowski 1995). We calculated 95% confidence intervals of nest survival by using the delta method (Powell 2007). We analyzed data from 2006 and 2007 together because we were interested in the overall effect of *S. invicta*, not annual differences, and we did not have sample sizes large enough to analyze each year independently. We used the 95% confidence intervals to compare estimates of nest survival and daily survival rate under the two treatments.

RESULTS

We monitored 71 nests, 44 of the White-eyed Vireo and 27 of the Black-capped Vireo. For the White-eyed Vireo, nest survival over the entire nesting cycle was 21% higher for treatment nests than for control nests; the estimate of nest survival for treatment nests was outside of the 95% confidence interval for control nests (Table 1). For the Black-capped Vireo, nest survival was 6% higher for treatment nests than for control nests, but the estimate for treatment nests was within the 95% confidence interval for control nests (Table 1).

TABLE 1. Daily survival rate and nest survival over one nesting cycle of White-eyed and Black-capped Vireo protected or not from predation by the imported red fire ant, estimated with the program MARK. Nests were monitored on private land in central Texas in 2006 and 2007.

	Daily survival rate	95% CI	Nest survival	95% CI	*n*
White-eyed Vireo					
Control	0.917	0.880–0.944	0.096	0.0–0.239	26
Treatment	0.957	0.927–0.976	0.308	0.123–0.493	18
Black-capped Vireo					
Control	0.916	0.861–0.951	0.073	0.0–0.262	14
Treatment	0.935	0.886–0.964	0.132	0.0–0.347	13

DISCUSSION

We found that preventing *S. invicta* from preying on nests increased nest survival of the White-eyed but not the Black-capped Vireo by more than the 10% we suggested might have a biologically significant effect on populations (Powell et al. 1999, Donovan and Thompson 2001). Because our experiment was manipulative, the results suggest White-eyed Vireo nest failure due to *S. invicta* occurred in addition to failure from other nest predators. *Solenopsis invicta* was likely a primary cause of nest failure because this species alone accounted for 20% of nest failures we observed. Anecdotally, we observed *S. invicta* preying on Black-capped Vireo nestlings in two control nests. We did not observe *S. invicta* trapped in the Tanglefoot pest barrier at treatment nests, suggesting the chemical barrier of the Spiral Wrap prevented the ant from preying on nests. Stake and Cimprich (2003) reported *Solenopsis* spp. in 11% of Black-capped Vireo nests on Fort Hood, Texas; Twedt et al. (2001) reported it in 10% of songbird nests on cottonwood (*Populus* spp.) plantations in Louisiana and Mississippi. Our estimates of nest survival for Black-capped Vireos protected from *S. invicta* predation are within previous estimates (Kostecke et al. 2005), suggesting the treatment did not deter or attract other nest predators. Although our treatment method acted as a barrier to all ant species, in our study area *S. invicta* was the most abundant ant, detected at 83% of ant-sampling stations near songbird nests (Campomizzi 2008), suggesting it poses the greatest threat to songbirds.

Interestingly, preventing nest predation by *S. invicta* caused a substantial increase in nest survival of the White-eyed but not the Black-capped Vireo. Both species nested in the same area, but small-scale nest-site characteristics may differ enough to enable other nest predators to attack Black-capped Vireo nests more frequently than White-eyed Vireo nests. White-eyed Vireos initiated their first nest of the season about 2 weeks earlier than Black-capped Vireos, potentially reducing their exposure to nest predators that are more active later in the breeding season. By using behavioral cues we found Black-capped Vireo nests easier to locate than White-eyed Vireo nests, suggesting visual predators may find Black-capped Vireo nests easier to locate as well. These spatial, temporal, and behavioral differences between the two species may suggest why the Black-capped Vireo's nest survival was lower. Increased predation on Black-capped Vireo nests by other predators would decrease the relative effect of *S. invicta*, explaining why for the Black-capped Vireo survival of treatment nests was not substantially higher than that of control nests.

The potential effects of *S. invicta* on nest survival of songbirds are substantial because of this ant's extensive and expanding worldwide range (Morrison et al. 2004, Sutherst and Maywald 2005). If our results are representative of the ant's effects, then nest survival of susceptible songbirds may be negatively affected over the 3.2×10^8 ha *S. invicta* currently occupies in the United States (USDA 2007). We expect the effects of *S. invicta* on songbirds' nest survival to vary spatially and temporally with the ant's distribution and abundance patterns, the complex interactions of nest-predator assemblages (Beauchamp et al. 1996, Jackson 2001, Schmidt et al. 2001), and other causes of nest failure.

Suppression of *S. invicta* in the breeding habitat of susceptible songbirds of conservation concern (i.e., threatened, endangered, locally rare) may be a management action to consider to increase the birds' productivity. Numbers of *S. invicta* may be suppressed through integrated pest management (Drees and Gold 2003, Pereira 2003). Our methods excluded *S. invicta* from nests, altering only the ant's function as a nest predator, not its numbers. Further research is needed because suppression of *S. invicta* in songbirds' breeding habitat may have ecological consequences on the nest-predator assemblage, such as an increase in snake or rodent populations, that are not yet realized.

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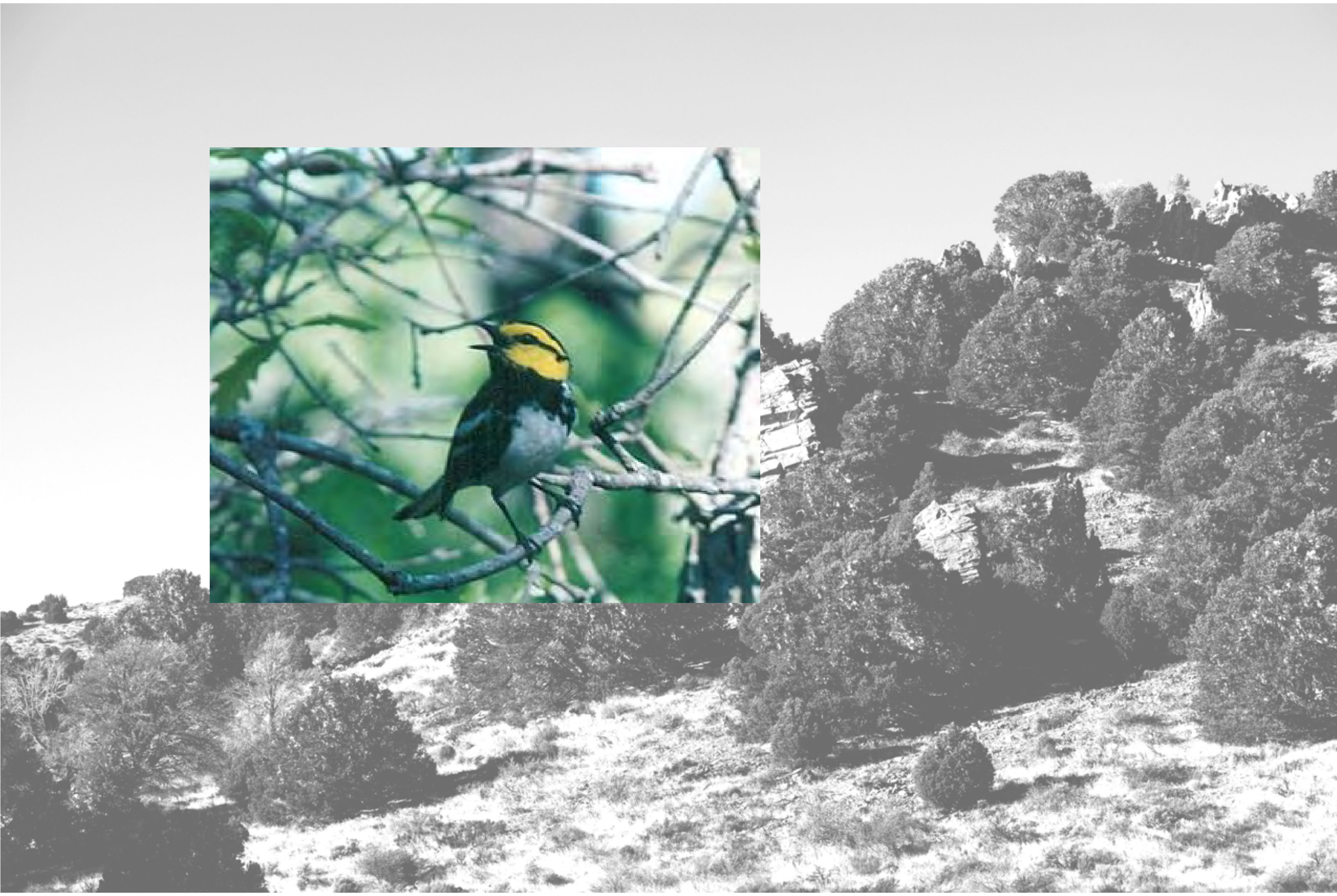
Appendix H

Golden-cheeked warbler habitat availability



GOLDEN-CHEEKED WARBLER HABITAT AVAILABILITY

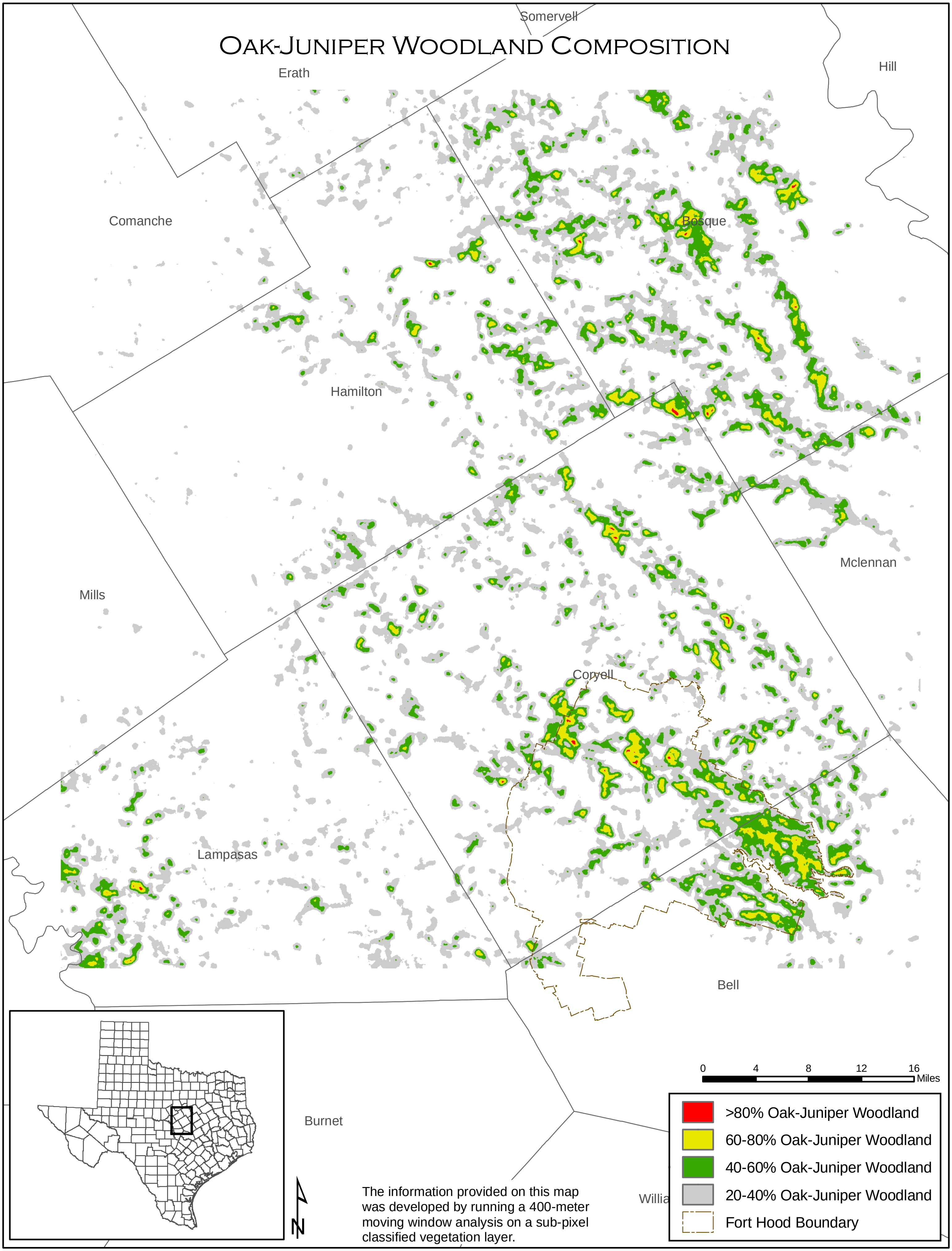
TEXAS A&M | Institute of Renewable Natural Resources

LANDINFO
LAND INFORMATION SYSTEMS

ABSTRACT

In an effort to determine habitat availability for Golden-cheeked warblers in Texas, we use a Landsat map to identify locations and size of oak-juniper woodland in Hamilton, Lampasas, Bosque, Coryell, and Bell counties, Texas. We compared Golden-cheeked warbler presence/absence data collected with habitat composition classes to evaluate which classes we were likely to locate Golden-cheeked warblers. Our descriptive statistics indicate that as composition class increased (>40 to >60% oak-juniper woodland) the frequency of point count stations surveyed with Golden-cheeked warblers increased.

OAK-JUNIPER WOODLAND COMPOSITION



The information provided on this map was developed by running a 400-meter moving window analysis on a sub-pixel classified vegetation layer.

METHODOLOGY AND SUMMARY STATISTICS

We used a Landsat map prepared by the University of Texas at Austin Center for Space Research (CSR) as a base layer for creating our map. CSR used subpixel classification to identify the vegetation composition across the study area. Our base map identifies all locations of oak-juniper woodland, the cover type used by golden-cheeked warblers (*Dendroica chrysoparia*). In this base map, each pixel is identified as oak-juniper woodland or another vegetation type. Our objective was to identify areas where ≥40%, ≥60%, and ≥80% of the area is composed of oak-juniper woodland. We used this range of oak-juniper composition percentages to maximize the information available for decision support, and we based these ranges on the available research regarding golden-cheeked warbler habitat vegetation characteristics.

We conducted ArcGIS™ Spatial Analyst Neighborhood Statistics procedure using the oak-juniper base map to identify areas with the 3 specified levels of oak-juniper composition. Using a 400-m radius window around each individual pixel, this procedure determines the proportion of pixels within that window composed of oak-juniper woodland. The resulting value for proportion of oak-juniper composition is assigned to each individual focal pixel. We used this method of classification to identify areas of ≥40%, ≥60%, and ≥80% oak-juniper woodland composition. The resulting maps show the locations of all areas in the study region with ≥40%, ≥60%, and ≥80% oak-juniper woodland composition.

In order to asses the utility of these maps for predicting golden-cheeked warbler occurrence, we validated the maps at each of the 3 levels of percent oak-juniper woodland composition, using data collected during 2006. Maps at the ≥40%, ≥60%, and ≥80% composition criteria were compared with data on golden-cheeked warbler occupancy at survey points within oak-juniper woodland within habitat at each percent classification. We estimated that in locations where oak-juniper woodland composition is ≥40%, the chance of not finding a golden-cheeked warbler is about 30%. We estimated that in locations where oak-juniper woodland composition is ≥60%, the chance of not finding a golden-cheeked warbler is about 15%. Therefore, the chance of finding a golden-cheeked warbler is greater in areas where oak-juniper composition is ≥60%. We were unable to make an accurate estimate for locations where oak-juniper woodland composition is ≥80% because we had only 3 survey stations within the ≥80% range. At ≥80% oak-juniper woodland composition, golden-cheeked warblers were detected at 2 out of the 3 survey stations.

We quantified the number of oak-juniper patches and the area covered by patches. Each patch is a continuous area with oak-juniper woodland composition. We summarized patches by size classes for ≥40%, ≥60%, and ≥80% levels of oak-juniper woodland composition for areas off of Fort Hood (Table 1) and for the entire study area (Table 2). Additionally, we evaluated the frequency of warbler detections across surveyed stations by habitat composition class for 2 patch sizes (Tables 3 – 4) as well evaluating the distribution of detections for all survey points across all habitat composition classes (Table 5).

Table 1. The number of patches and total acres for each patch size class and percent oak-juniper composition category for patches off of Fort Hood in Hamilton, Lampasas, Bosque, Coryell, and Bell counties.

	Percent of area composition within a 400m radius					
	>40% composition		>60% composition		>80% composition	
Patch Size (acres)	No. of patches	Total acres	No. of Patches	Total acres	No. of patches	Total acres
0 – 10	364	605	137	358	16	41
10 – 50	176	4,635	65	1,781	11	236
50 – 100	97	7,073	36	2,557	1	80
100 –250	134	21,138	27	4,058	0	0
250 – 500	62	21,989	9	2,943	0	0
>500	45	53,945	8	4,911	0	0
Totals	878	109,385	282	16,608	28	357

Table 2. The number of patches and total acres for each patch size class and percent oak-juniper composition category for all patches in Hamilton, Lampasas, Bosque, Coryell, and Bell counties.

	Percent of area composition within a 400m radius					
	>40% composition		>60% composition		>80% composition	
Patch Size (acres)	No. of patches	Total acres	No. of Patches	Total acres	No. of patches	Total acres
0 – 10	402	634	190	424	25	53
10 – 50	194	5,049	88	2,412	16	374
50 – 100	105	7,644	47	3,402	1	80
100 –250	147	23,127	37	5,856	0	0
250 – 500	72	25,762	11	3,662	0	0
>500	60	89,084	14	11,566	0	0
Totals	980	151,300	387	27,322	42	507

2006 SURVEY DATA SUMMARY

- We detected golden-cheeked warblers at 73% (n = 59) of point count stations surveyed in >40% oak-juniper composition in patches >250 acres.
- We detected golden-cheeked warblers at 80% (n = 15) of point count stations surveyed in >60% oak-juniper composition in patches >250 acres.
- We detected golden-cheeked warblers at 75% (n = 69) of point count stations surveyed in >40% oak-juniper composition in patches >100 acres.
- We detected golden-cheeked warblers at 88% (n = 24) of point count stations surveyed in >60% oak-juniper composition in patches >100 acres.

Table 3. The number of point count stations surveyed and number of stations with golden-cheeked warbler (GCWA) detections in >40% and > 60% oak-juniper woodland in patches >250 acres.

	Patches > 250 acres					
	> 40% oak-juniper woodland			>60% oak-juniper woodland		
Year	Stations Surveyed	Stations w/GCWA	% w/GCWA	Stations Surveyed	Stations w/GCWA	% w/GCWA
2003	56	25	45	0	0	0
2004	131	66	50	7	5	71
2005	60	17	28	5	1	20
2006	61	42	69	15	12	80

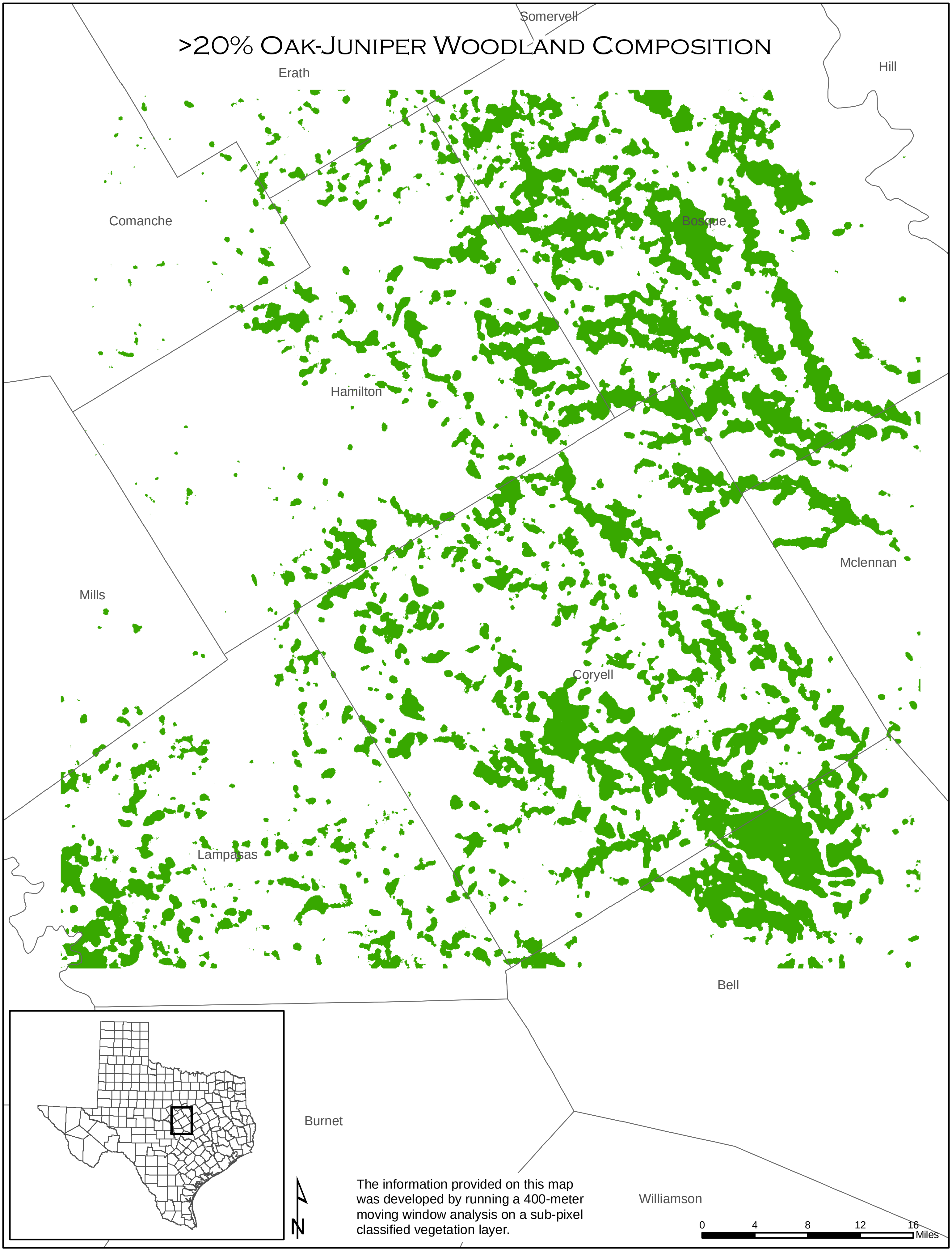
Table 4. The number of point count stations surveyed and number of stations with golden-cheeked warbler (GCWA) detections in >40% and > 60% oak-juniper woodland in patches >100 acres.

	Patches > 100 acres					
	> 40% oak-juniper woodland			>60% oak-juniper woodland		
Year	Stations Surveyed	Stations w/GCWA	% w/GCWA	Stations Surveyed	Stations w/GCWA	% w/GCWA
2003	67	34	51	6	4	67
2004	150	75	50	20	18	90
2005	90	23	26	5	1	20
2006	78	52	67	28	24	86

Table 5. Distribution of point count stations by percent composition. The total number of point count stations surveyed each year are categorized by detection (yes) or non-detection (no) of golden-cheeked warblers. Point count surveys are totaled for all years.

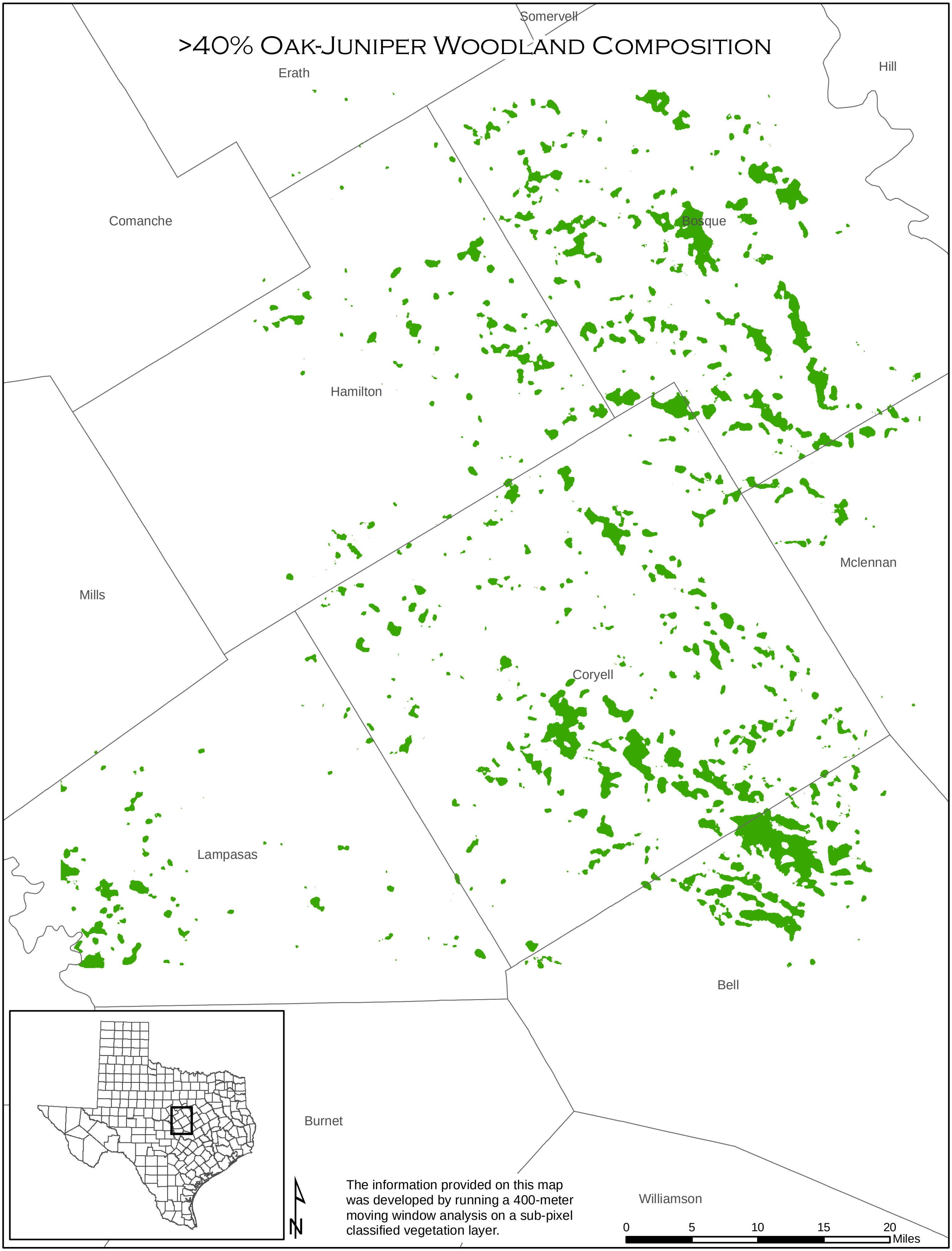
	2003			2004			2005			2006			Total	
Composition	Yes	No	Total	Yes	No	Total	Yes	No	Total	Yes	No	Total	Yes	No
<20	1	134	135	4	60	64	0	75	75	2	30	32	7	299
20 – 40	29	135	164	38	117	155	12	99	111	23	72	95	102	423
40 – 60	24	41	65	72	87	159	16	66	82	33	20	53	145	217
60 – 80	11	3	14	15	5	20	9	9	18	24	4	28	59	21
>80	0	0	0	0	0	0	0	0	0	2	1	3	2	1
Totals	65	313	378	129	269	398	37	249	286	84	127	211	315	958

>20% Oak-Juniper Woodland Composition



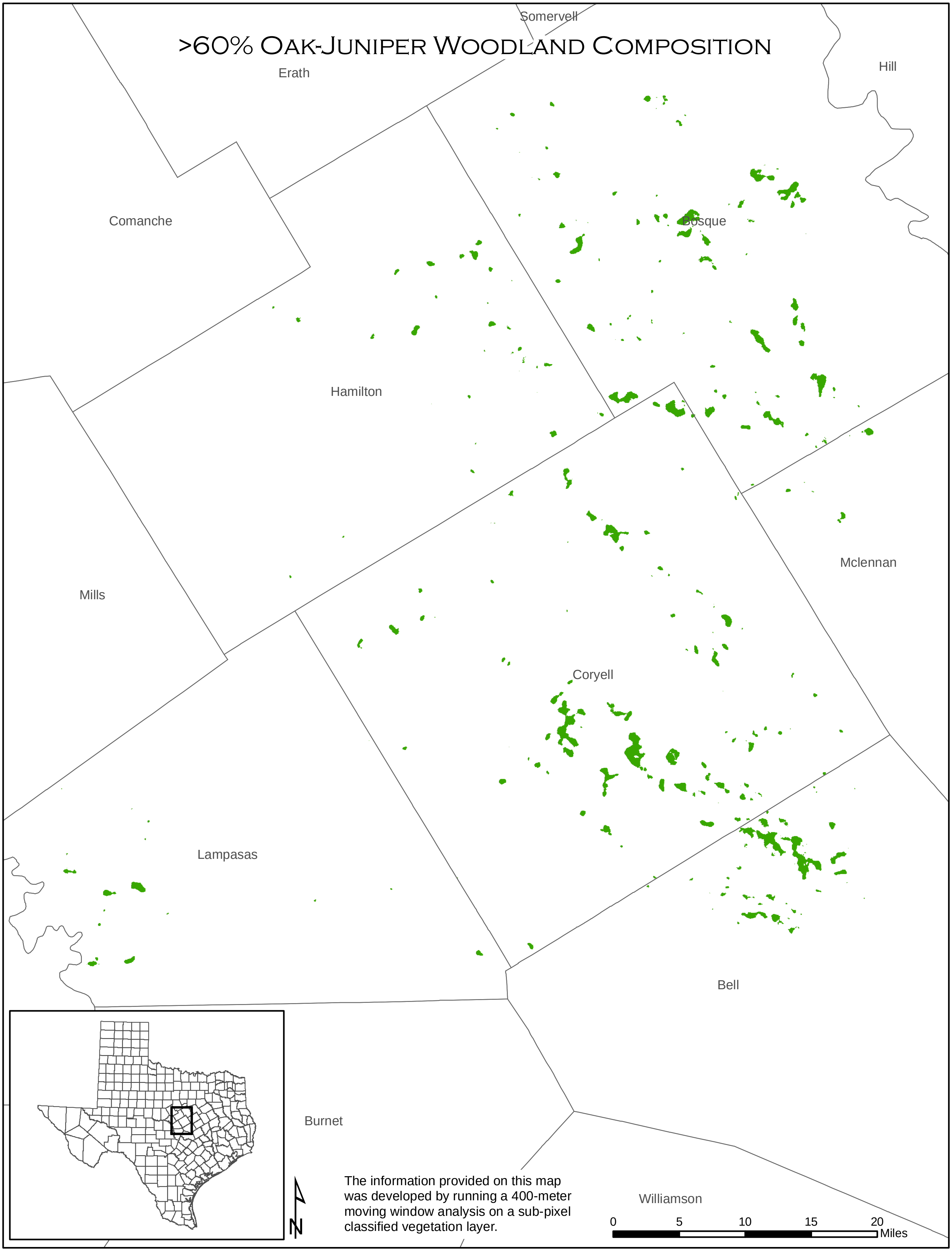
The information provided on this map was developed by running a 400-meter moving window analysis on a sub-pixel classified vegetation layer.

>40% Oak-Juniper Woodland Composition



The information provided on this map was developed by running a 400-meter moving window analysis on a sub-pixel classified vegetation layer.

>60% Oak-Juniper Woodland Composition



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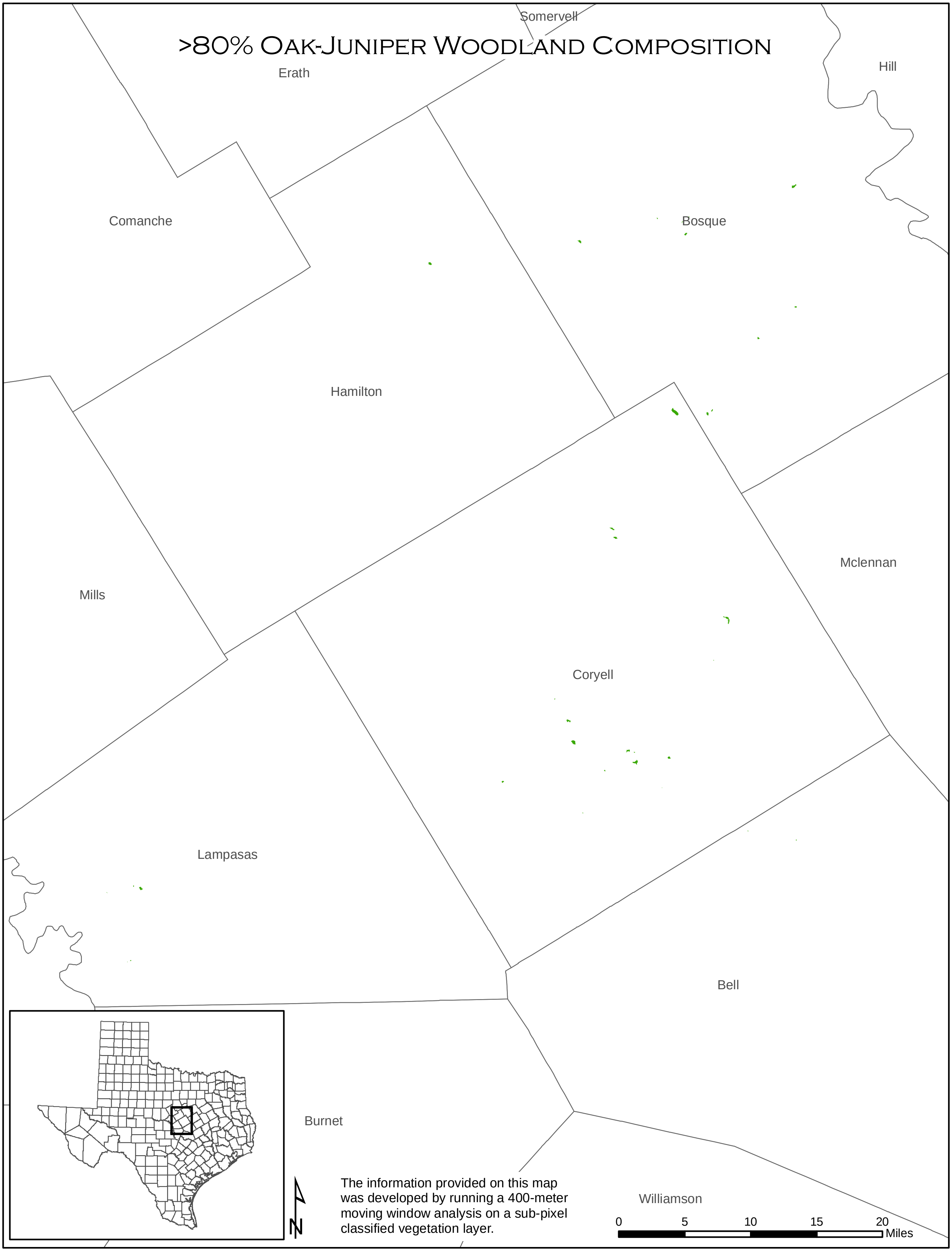
>80% OAK-JUNIPER WOODLAND COMPOSITION



The information provided on this map was developed by running a 400-meter moving window analysis on a sub-pixel classified vegetation layer.

Appendix I

Monitoring golden-cheeked warblers on private lands in Texas

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Management and Conservation Note

Monitoring Golden-Cheeked Warblers on Private Lands in Texas

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1 **ABSTRACT** A majority of North American breeding habitat for neotropical migrants exists on private lands, requiring monitoring strategies focused on habitat in these private holdings. We outline study designs and protocols using repeated presence–absence surveys across a gradient of patch sizes to develop a range-wide monitoring program for the endangered golden-cheeked warbler (*Dendroica chrysoparia*) in Texas, USA. We surveyed 200–400 point-count locations across approximately 30 private properties annually from 2005 to 2008. We used data from our surveyed patches ($n = 147$) and the Ψ (occupancy), p (detection), and $\gamma = 1 - \varepsilon$ parameterization to estimate patch dynamics and associated detection probabilities for golden-cheeked warblers. Patch size had a strong association with patch occupancy, and all patches >160 ha were predicted to be occupied. We found no evidence that large golden-cheeked warbler populations located on public lands in the vicinity of our study area influenced occupancy dynamics. We conducted simulations across a range of detection probabilities to evaluate potential sample sizes for both standard- and removal-based occupancy modeling. Simulations using parameter estimates from our analysis indicated that removal-based sampling is superior to standard sampling. Based on our results, surveying golden-cheeked warbler presence in oak–juniper (*Quercus–Juniperus*) patches under a removal modeling framework should be considered as one alternative for range-wide monitoring programs because patch-level monitoring would be necessary to estimate proportion of range occupied. Large contiguous patches are rare across the species' range; hence, conservation and management of the mosaic of smaller patches within a landscape context would be required for maintaining species viability. Thus, we recommend the identification of areas where smaller, contiguous patches represent a significant portion of the available habitat within the local landscape and targeting these areas for habitat maintenance and improvement.

KEY WORDS breeding range, *Dendroica chrysoparia*, fragmentation, golden-cheeked warbler, habitat loss, patch area, patch occupancy, Recovery Credit System.

The golden-cheeked warbler (*Dendroica chrysoparia*) is a federally endangered neotropical migratory passerine with a known breeding range across about 35 counties ($\geq$95% private ownership) in central Texas, USA (Fig. 1; Pulich 1976, Deboer and Diamond 2006, Magness et al. 2006). Within the warblers' breeding range, mature oak (*Quercus* spp.)–Ashe juniper (*Juniperus ashei*) woodlands provide foraging habitat, nesting cover, and shredded bark used as nesting substrate (Pulich 1976, Ladd and Gass 1999). However, declines in oak–juniper woodlands (United States Fish and Wildlife Service 1992) and the small percentage of breeding habitat found on public lands (<5% of total area within the breeding range) requires that conservation planning for the warbler must incorporate habitat on private lands. Understanding the combined impacts of habitat loss and fragmentation, as well as the impacts of private lands on species distribution and demography, is vital to recovery (United States Fish and Wildlife Service 1992).

Several models of habitat distribution of the golden-cheeked warbler have been constructed to guide conservation efforts (e.g., DeBoer and Diamond 2006, Magness et al. 2006). Habitat delineations are usually based on presence or absence data collected during point-count surveys on private and public lands (Wahl et al. 1990). However, current habitat models have been based on a limited number of survey points across the species' range ($n = 49$, DeBoer and Diamond 2006; $n = 202$, Magness et al. 2006). Although attempts have been made to predict distribution of warbler habitat, recent studies have relied on short-term (single season) and limited-visit ($\leq$2) surveys to establish species presence for identifying general habitat metrics (Wahl et al. 1990, DeBoer and Diamond 2006, Magness et al. 2006).

Limited data exist for modeling golden-cheeked warbler demographics across the species' range, with most data derived from studies on Ft. Hood, Coryell County (Fig. 1; Anders 2000, Alldredge et al. 2004, Anders and Dearborn 2004, Baccus et al. 2007, Peak 2007). Recent efforts to quantify golden-cheeked warbler population size have been based on a combination of density estimates from approximately 30 years ago (Pulich 1976), from the intensively managed Ft. Hood populations (Jettj et al. 1998), or from a small number of transects ($n = 11$, Wahl et al. 1990). Density estimates are then combined with aforementioned estimates of available breeding habitat (e.g., Wahl et al. 1990, DeBoer and Diamond 2006, Magness et al. 2006) to estimate population size (Rappole et al. 2005). These derived population estimates remain unreliable for use in the range-wide management and conservation of the species. 2

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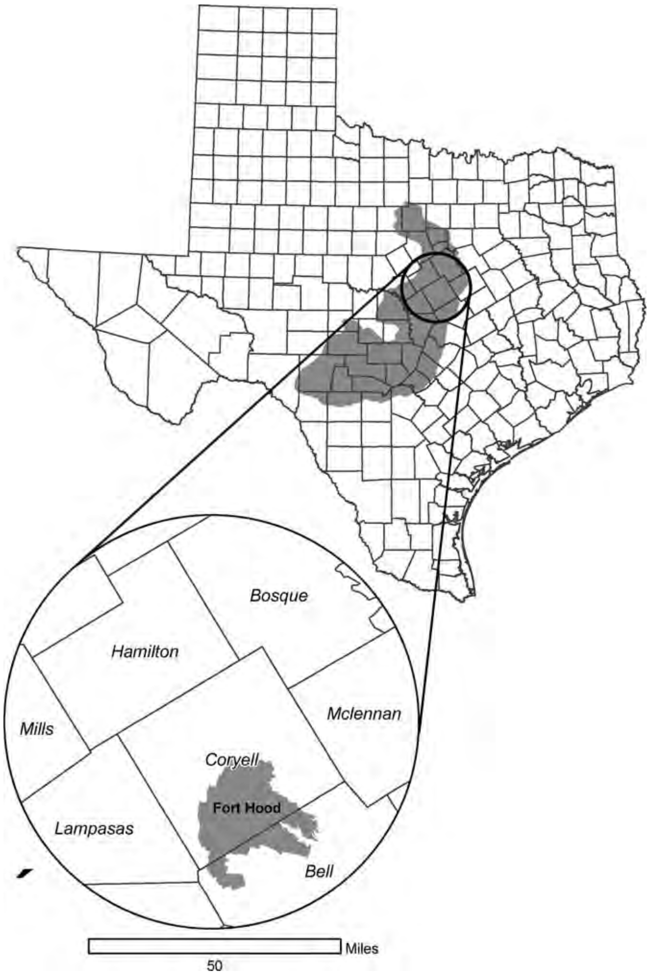

Figure 1. Study area where golden-cheeked warbler habitat patch surveys were conducted during 2006–2008 relative to Ft. Hood and the breeding range of the golden-cheeked warbler in central Texas, USA.

Periodic assessments of changes in populations are necessary to understand implications of management (Pollock et al. 2002, Nichols et al. 2008). Successful monitoring programs for avian species require that managers use reliable estimators for rapidly detecting population trends and measuring the magnitude of such changes (Williams et al. 2002). Our objective is to test and standardize population monitoring protocols for golden-cheeked warblers in the vicinity of Ft. Hood, Texas using repeated presence–absence surveys and make recommendations for expanded monitoring efforts throughout the breeding range. In addition, we evaluated potential sampling approaches for implementing range-wide distribution and monitoring surveys for golden-cheeked warblers.

STUDY AREA

We conducted our research centered on the Leon River watershed in the Lampasas Cut Plains and Cross Timbers and Prairies region of central Texas (Fig. 1; Gould 1975). This region was characterized by ecological sites of steep adobe, low stony hills, and loamy bottomlands. Dominant tree species included Ashe juniper, oaks, ash (*Fraxinus* spp.), and elm (*Ulmus*) species, as well as pecan (*Carya illinoinensis*) and hackberry (*Celtis laevigata*). Approximately 84% of the area was mixed-use agriculture, consisting of rangeland and croplands with ongoing urban development near the Interstate 35 corridor. Approximately 13% of our study area was mature oak–juniper woodland, primarily in patches <18 ha. Patches >18 ha made up >80% of the total mature oak–juniper woodland (Butcher 2008). 3

METHODS

Avian studies often define sampling sites as locations where measurements of presence–absence or abundance are collected (e.g., point-count stations; Ralph et al. 1995, Nichols et al. 2000, Thompson 2002). However, sites defined by such criteria are not likely to be a landscape unit of ecological importance to the species of interest. Given the endemism of golden-cheeked warblers to patches of oak–juniper woodlands (Pulich 1976, Ladd and Gass 1999), we defined oak–juniper patches as being the operational sampling unit for considering occupancy. In addition, this scale is likely the same scale at which measurements of demography (abundance, survival, productivity) would be relevant from a recovery or management perspective (Gilpin and Hanski 1991, MacKenzie and Royle 2005, MacKenzie et al. 2006). 4

We delineated oak–juniper patches using unsupervised classifications of Landsat Thematic Mapper images with ERDAS IMAGINE 9.2. We used Landsat images representing summer, spring, and autumn to distinguish between evergreen and deciduous trees to separate mixed oak–juniper patches from continuous oak patches (e.g., riparian areas). We used ArcMap 9.2 to locate point-count stations within identified juniper–oak forest patches. We estimated patch area (ha) using the VLATE 1.1 (Lang and 5 Tiede 2003) extension of ArcMap. We estimated a proximity index for each sample patch based on the size and distance of all patches having edges within a specified search radius (400 m; Magness et al. 2006) of the focal patch (Gustafson and Parker 1992). We assessed classification accuracy following the descriptive technique described by Congalton (1991). We visited 161 systematically placed reference plots (30 × 30 m) to determine vegetation cover and composition. We defined mixed juniper–deciduous forest as forest with >50% canopy cover, of which ≥10% was juniper and ≥10% was deciduous species. The overall accuracy of the unsupervised classification was 78%. The probability that a reference point was correctly classified as juniper–deciduous forest was 79% and probability that a reference point was correctly classified as other was 75%.

Our interest was in surveying patches of habitat on private lands within the region surrounding known populations at Ft. Hood, and our sampling frame consisted of accessible private properties. We based selection of patches for surveying on availability of potential golden-cheeked warbler habitat, which we delineated using our habitat classification schema. For each accessible property (n = 30), we systematically distributed point-count stations throughout available oak–juniper woodland patches. The number of surveyed point-count stations and patches surveyed varied over time depending on property access, ranging between 200 to 400 total point-count locations. Each point-count location was separated by >250 m from other stations. In addition, surveyors listened during movements between points for the warbler as well as several additional species of management interest known to occupy our study area. During pilot field work evaluating data collection and survey methodology (2003–2005), we surveyed each point-count location 3 times under a standard occupancy design (MacKenzie et al. 2002, 2006). Preliminary analysis of 2003–2005 data indicated that the number of repeated surveys should be increased to 6 surveys to better evaluate variation in warbler detection rates across the breeding season (B. A. Collier, Texas A&M University, unpublished data). Given results of our pilot analysis (2003–2005), we then used data collected from 2006 to 2008 for the present analysis.

We transitioned our 2008 survey efforts to a removal modeling approach where 2 observers traversed between point-count stations within a patch simultaneously but independently (MacKenzie and Royle 2005) because our pilot analysis indicated that detection rates where high ($>50\%$) and declined in conjunction with changes to warbler reproductive phenology (e.g., territory settling, incubation, nesting feeding; Ladd and Gass 1999). According to the removal design, an observer's survey ended when he or she made a positive detection. However, because observers operated independently, the other observer continued to survey within the patch until he or she made a positive warbler detection or until the patch was completely traversed and surveyed. Thus, for each survey occasion, possible encounter histories for detections by one or both observers were 10, 01, 11, or 00 (detection by first observer only, detection by second observer only, detection by both observers, no detections, respectively). If no positive detections were made during a survey, we conducted additional surveys following the above protocol for a maximum of 6 presence–absence surveys (3 visits by 2 observers over 3 weeks) for any patch, equivalent to the number of sample surveys conducted during 2006–2007. Our survey techniques were consistent from 2006 to 2008 with observers randomly allocated to sample locations, point-count stations traversed during morning hours (0600–1100), and repeated visits to a patch separated by <7 days.

Selection of habitat patches for surveying did not require presence of golden-cheeked warblers, only presence of predicted warbler habitat. We avoided site selection bias in our estimates of occupancy because we did not have preexisting knowledge of the potential occupancy state (MacKenzie and Royle 2005). Property ownership distribution was variable; therefore, in some cases we did not have access to the entirety of a patch. However, we assumed that habitat and warbler distribution was not influenced by nonbiological boundaries (e.g., property lines) so all points within a patch were equally likely to have warblers within the vicinity.

We used data from our surveyed patches and the Ψ (occupancy), p (detection), and $\gamma = 1 - \varepsilon$ parameterization in PRESENCE (MacKenzie et al. 2006) to estimate patch occupancy and associated detection probabilities for golden-cheeked warblers. We defined the primary sampling occasions as years (3 yr, 2006–2008) and our secondary sampling occasions were the repeated patch visits that occurred during the warbler breeding season (≤ 6 visits from 2006 to 2008; Mar–Jul). For our candidate models, we focused modeling of detection on session-dependent covariates associated with the sampling process. We expected that as the season progressed, detections would decline as breeding activities shifted from territory establishment and mating to other reproductive activities (e.g., nesting, feeding nestlings; Ladd and Gass 1999), and based on previous analyses of earlier data (M. L. Morrison, Texas A&M University, unpublished data), we regarded this parameter as fixed in all our models. Thus, we modeled survey week as a session-dependent covariate and we defined 15 March as the beginning of week 1, because this date is when warblers began arriving at our study area.

Our breeding survey observations suggested that golden-cheeked warblers occupied a wide variety of patches ranging from 1 ha to $>1{,}000$ ha. Given the dependence on oak–juniper patches, as the size of habitat patches increased we expected some threshold level (Lindenmayer and Luck 2005) of patch area at which $p(\Psi) = 1$. Because management for golden-cheeked warblers has been concentrated on a few public holdings (e.g., Ft. Hood; Anders and Dearborn 2004, Baccus et al. 2007) under the expectation that birds would be recruited into surrounding areas from these source populations (e.g., United States Fish and Wildlife Service 2001), we modeled distance from Ft. Hood as a predictor for warbler patch occupancy. We used an information theoretic approach to model selection and assessed model strength based on Akaike's Information Criterion adjusted for small sample size (AIC_c) and Akaike weights (w_i; Burnham and Anderson 2002). When model selection uncertainty occurred, we used multimodel inference and provide model-averaged estimates of parameters (Burnham and Anderson 2002). 6

Using parameter estimates from our best-fitting model, we estimated average occupancy for surveyed patches of habitat. Based on our average occupancy estimates (0.495) we used R (R Core Development Team 2008) to simulate potential sample sizes necessary for a range of estimated detection probabilities and repeated survey frequencies. We assumed a fixed variance of the occupancy parameter [$\mathrm{var}(\hat{\Psi};) = 0.05$] for all simulations, and we evaluated sample requirements across a range of minimum (standard design) and maximum (removal design) repeated visits. Based on our results (see below) we simulated sample requirements over the range of detection probabilities garnered from the first 2 months of the breeding season when birds were actively engaged in reproductive activities (e.g., territory establishment). We used the sample size formulas of MacKenzie and Royle (2005) for both standard and removal modeling designs to determine optimal sample sizes under both data collection methods that would be optimal for monitoring golden-cheeked warbler distribution across their range.

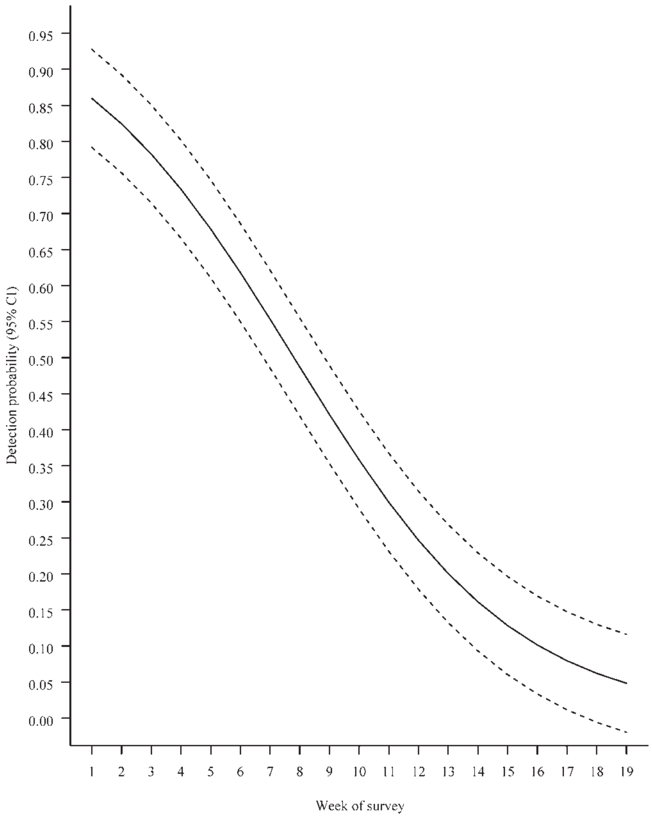

Figure 2. Predicted weekly (15 Mar = week 1) detection probabilities for golden-cheeked warbler habitat patch occupancy surveys conducted in central Texas, USA, 2006–2008.

RESULTS

We monitored 147 habitat patches for ≥ 1 breeding season between 2006 (72 patches), 2007 (94 patches), and 2008 (39 patches). Patch sizes averaged 45 ha (SD = 111) and ranged from 0.54 ha to 1,043 ha. Our modeling of detection probabilities showed a consistent pattern of declining detections as week of the year progressed (Fig. 2). Secondary, occasion-specific detection probabilities associated with our robust design were high for each period and declined as expected over the course of the season. The one exception to this finding was in 2008 when our removal modeling approach with double observers reduced the number of required visits to sites, which reduced the amount of the breeding season we spent surveying (Fig. 3). Our models including patch area as a main or interactive effect had more support than models for occupancy as a function of other variables (Table 1). Model-averaged predictions across the range of patch area showed a increase in patch occupancy, reaching the threshold [prob(Ψ) = 1] at 160 ha in size (Fig. 4). We evaluated competing models for patch proximity and distance from Ft. Hood; however, none were supported in our analysis. Models using patch proximity (intercept = −0.1893 [SE = 0.198], slope = 0.001863 [SE = 0.00064]), or distance from Ft. Hood (intercept = 0.2604 [SE = 0.178], slope = 0.0000001 [SE = 0.00024]) as predictors for occupancy indicated that neither factor was likely impacting occupancy dynamics at
7 the scale we evaluated.

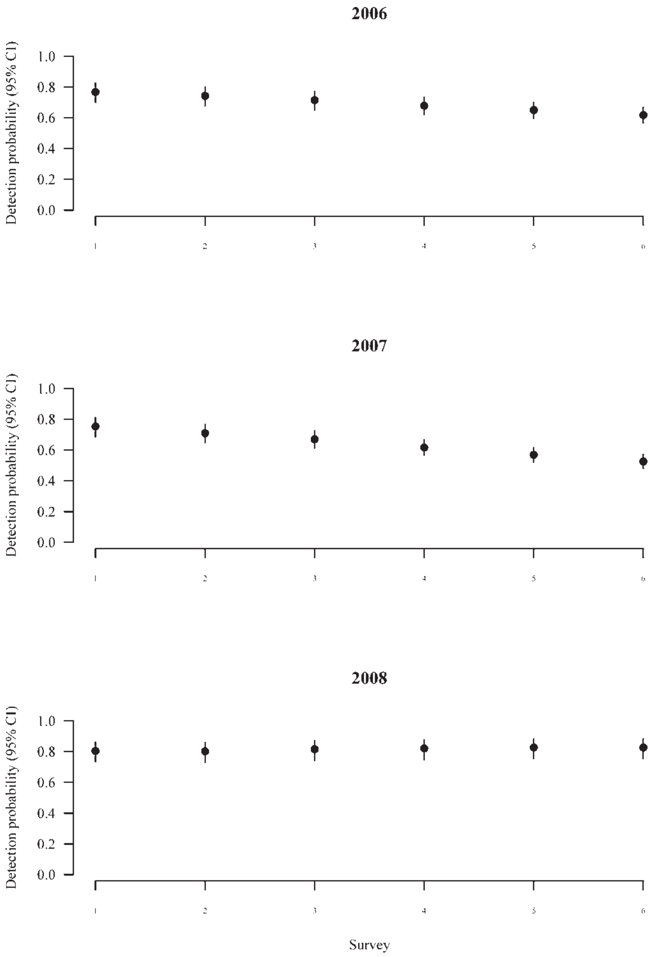

Figure 3. Detection probabilities across survey periods for golden-cheeked warbler habitat patch occupancy surveys conducted in central Texas, USA, 2006–2008.

Our simulation results showed that both removal and standard designs tended to converge on optimal numbers of patches to survey between 90 patches and 110 patches as detection probability increased (Fig. 5). For our simulated results, we found that to estimate a standard error equal to 0.05 for occupancy, the number of sites to survey was <120 as long as the maximum number of allowable visits was ≥ 4 for removal designs. For the standard design, the number of sites to survey was <150, assuming each site would be surveyed ≥ 3 times each season (Fig. 5). 9

DISCUSSION

Using an appropriately defined sampling unit, different monitoring programs for rare species such as the golden-cheeked warbler can be evaluated, and issues associated with low detections can be more adequately investigated rather than relying on complex statistical models (McDonald 2006, Royle 2006). High detection estimates suggest our choice of sampling units (i.e., the patch rather than points within a patch) was appropriate for evaluating our monitoring program's primary state variable of interest, patch occupancy (Bailey et al. 2004). Occupancy surveys developed for monitoring that treat survey points within a habitat patch as independent sampling units are likely pseudoreplicated

Table 1. Candidate models used to examine the effects of oak–juniper habitat patch area (Patch Area) and distribution (Proximity), distance from potential source populations on Ft. Hood (DistFtHood), and survey timing (vwk) on patch occupancy and detection of golden-cheeked warblers in the Leon and Bosque River watersheds of Texas, USA, 2006–2008. 8

Model notation	No. of parameters	-2LL	ΔAIC$_c$	w_i
p(vwk)Psi(PatchArea)Gamma(PatchArea)	6	831.727	0	1.00
p(vwk)Psi(Proximity)Gamma(PatchArea)	6	854.36	22.63	0
p(vwk)Psi(Ft. Hood)Gamma(PatchArea)	6	860.62	28.89	0
p(vwk)Psi(PatchArea)Gamma(t)	6	876.33	44.60	0
p(vwk)Psi(PatchArea)Gamma(Proximity)	6	885.94	54.21	0
p(vwk)Psi(PatchArea)Gamma(t.)	5	889.55	55.82	0
p(vwk)Psi(PatchArea)Gamma(Ft. Hood)	6	889.55	57.82	0
p(vwk)Psi(.)Gamma(t)	5	906.17	72.44	0
p(vwk)Psi(Ft. Hood)Gamma(t)	6	905.22	73.50	0
p(vwk)Psi(Proximity)Gamma(.)	5	912.18	78.46	0
p(vwk)Psi(.)Gamma(.)	3	984.08	146.35	0

(Hurlbert 1984). Point-count surveys are appropriate if the primary interest is in determining the relationship between local (e.g., within-patch) habitat conditions and bird presence (Lauver et al. 2002, Kroll et al. 2007). However, use of points as the sampling unit is not appropriate for long-term, range-wide monitoring because perturbations that affect individual dynamics operate at a scale much larger than the point and may be associated with processes associated with inter-patch territory establishment and territory success or failure. 10

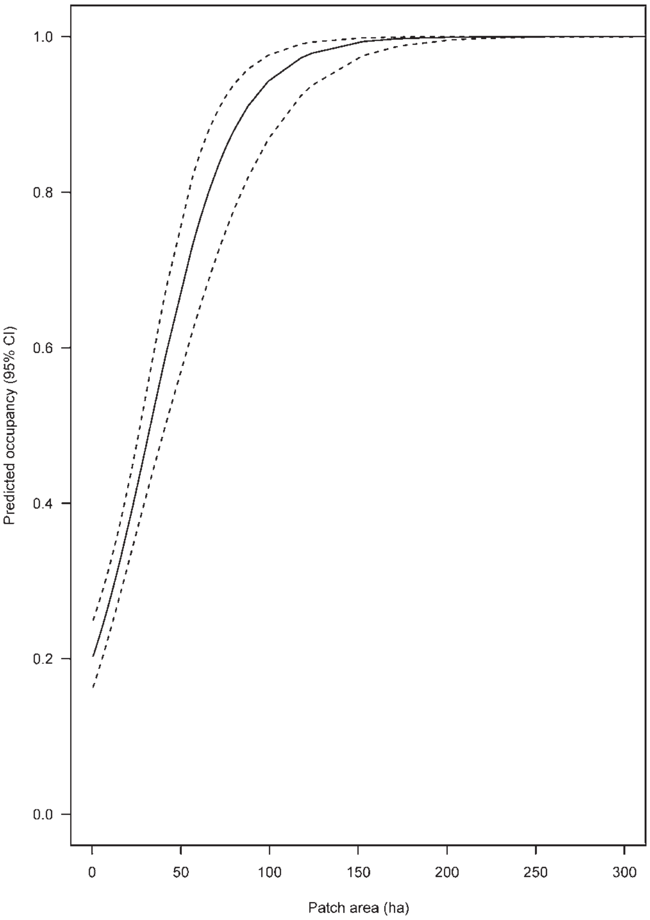

Figure 4. Predicted occupancy and associated 95% prediction interval for golden-cheeked warbler habitat patches surveyed in central Texas, USA, 2006–2008. Note the prediction range is truncated at 300 ha [p(Ψ) = 1] while maximum patch area surveyed was approximately 1,042 ha.

Using a random-changes occupancy model we expected no relationship between t and $t + 1$ in predictions of occupancy. The relationship between occupancy in time 1 and future sampling occasions was a function of the covariates used for predicting $\hat{\Psi};_1$, and $\hat{\gamma};_1$ and $\hat{\gamma};_2$ (MacKenzie et al. 2006). 11 That our expectation occupancy would be positively associated with patch area seems logical because positive relationships between occupancy, local abundance, and regional distribution are common patterns in population ecology (Gaston et al. 1997, He and Gaston 2000, Holt et al. 2002). Within larger patches that had high occupancy probabilities, increased focus should be put on estimating within-patch state variables (i.e., intra-patch distribution and movements, abundance, survival, and productivity). Given the wide range of patch sizes across the breeding range of Texas (present study, Ladd and Gass 1999), our results indicate that golden-cheeked warbler occupancy surveys should focus on patches ≤ 160 ha because 1) factors influencing presence would be most likely to influence smaller patches; 2) smaller patches represent a large proportion of the occurrence of available habitat on private lands across many of the species' recovery regions; and 3) smaller patches are caused by increasing fragmentation via land ownership changes across the breeding range and, hence, are more likely to be degraded over time due to less management (Sanders 2005). Providing incentives for habitat conservation on private lands is one option for reducing impacts of habitat fragmentation. A program that provides incentives to landowners for managing golden-cheeked warbler habitat is being implemented and evaluated in our study region (Recovery Credit System; Wilkins et al. 2009). 12

We did not find evidence that locally abundant populations on Ft. Hood influenced warbler distribution and occupancy on adjacent private land (proposed by Anders and Dearborn 2004, Baccus et al. 2007). We suggest several plausible explanations: 1) Ft. Hood is not serving as a source population for golden-cheeked warblers in the Leon River watershed; 2) dispersal of birds from Ft. Hood had already saturated our study area before our study began; 3) colonization or extinction processes occurred at a much finer resolution (within-patch or territories) than we evaluated; or 4) sampling patches transitioning into and

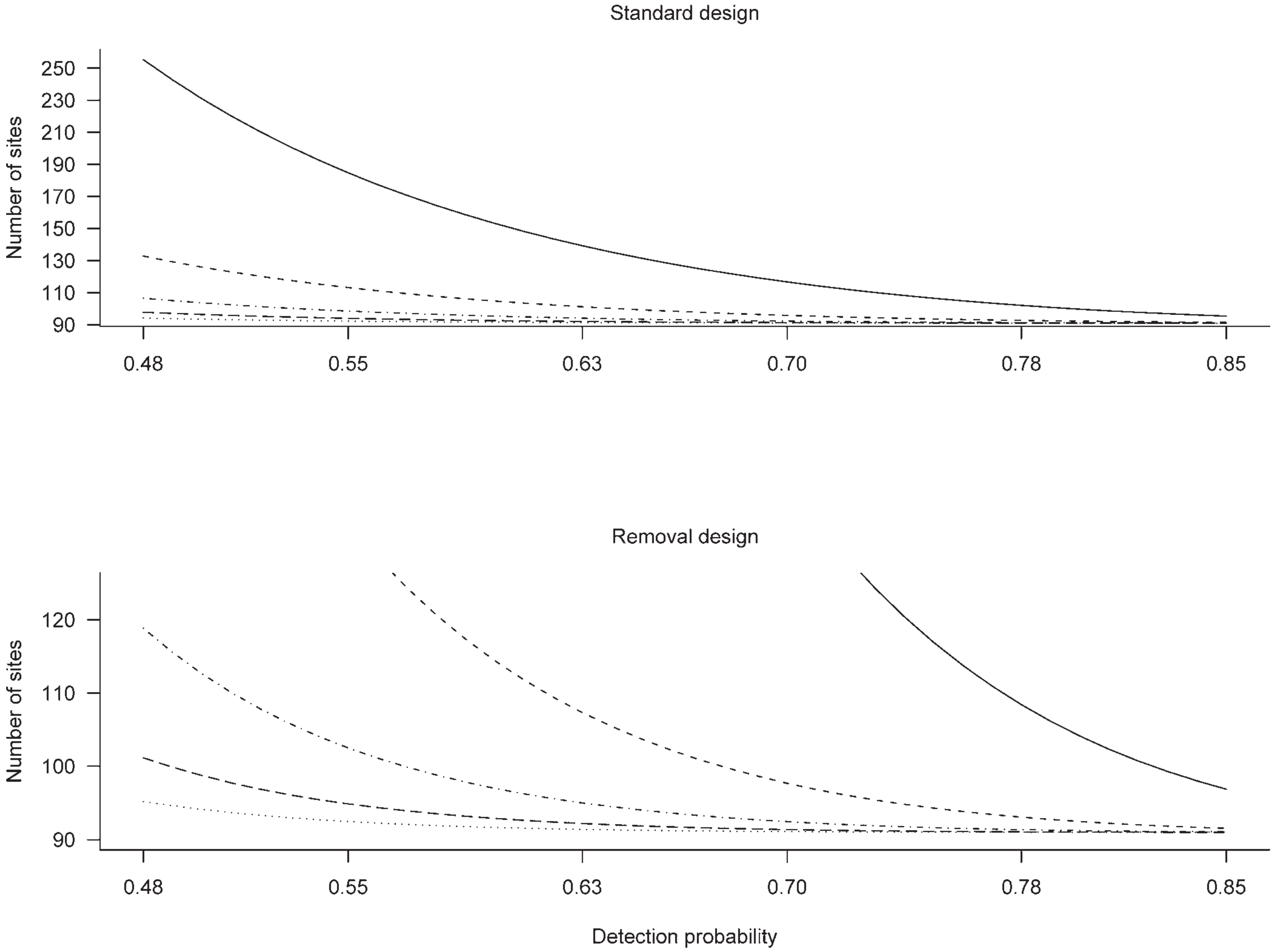

Figure 5. Simulated number of sites necessary to survey for golden-cheeked warblers under both standard and removal occupancy designs to attain a standard error of the occupancy estimate of 0.05 based on detection probabilities associated with the period 15 March to 15 May survey periods used in Texas, USA, during 2006–2008. We ran simulations across a range of *K* (no. of surveys) show by the solid line ($K = 2$) through the dotted line ($K = 6$). Under a standard design, *K* represents the minimum number of surveys necessary to attain the prespecified level of precision, while under the removal design, *K* represents the maximum number of surveys that would be required if presence was not confirmed.

out of our study based on property access restrictions simply inhibited our ability to accurately estimate these processes. Thus dynamic processes associated with colonization–extinction dynamics, if those are biologically interesting and meaningful for species recovery, would likely necessitate estimation based on intra-patch dynamics focused specifically on measurement within the utilized territory of the bird.

Issues associated with low detection probabilities can be more adequately investigated using sample survey (i.e., design-based) approaches rather than the less rigorous model-based inferences (Royle 2006). We surveyed for 19 weeks to estimate detection probabilities over the course of the breeding season and we likely violated the closure assumption (MacKenzie et al. 2006). However, because our intention was to fully evaluate a broad range of factors influencing distribution and occupancy, gaining knowledge of how detection varied temporally warranted expanding the survey period. Hence, our pilot survey results (B. A. Collier, unpublished data) and this study indicated that the temporal scale of surveying most influenced detection probability within the patch.

Our results concur with the general results from MacKenzie and Royle (2005) wherein removal designs were efficient in situations where occupancy exceeded 30%, regardless of detection probability (MacKenzie and Royle 2005). However, the number of surveys under a removal design is a random variable and can necessitate a greater number of overall surveys when detections do not occur, so there is an element of chance associated with removal design (MacKenzie and Royle 2005). One plausible approach to increasing efficiency under a removal design would be to incorporate a cost function that addresses costs associated with survey visits by individuals compared to multiple observers during the same survey event. Specifying this cost function is critical, because optimal sampling designs must include a consideration of available resources. For example, if multiple observers could survey a location concurrently, then logistical costs associated with survey effort could be reduced, while sample intensity (e.g., no. of sample surveys conducted) could be increased. Thus, while the researcher must be willing to repeatedly survey a site for up to n times under a removal design to gain the efficiency benefits (MacKenzie et al. 2006), simple changes in survey design could potentially offset any increased costs associated with removal models.

Management Implications

Future golden-cheeked warbler surveys should be conducted between 15 March and 1 May each year to take advantage of high detection probabilities during this period. In the context of recovery planning, monitoring and evaluating patch-occupancy dynamics should occur among smaller patches (<160 ha). Work within larger patches should focus on intra-patch distribution, patch level abundance and productivity. Future monitoring data following our design should be used for supporting approaches to minimize or mitigate against future habitat loss in highly vulnerable areas across the species' range. We suggest that the study design and analyses used herein for the golden-cheeked warbler would be applicable to other rare, woodland endemic species.

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Appendix J

Conspecific attraction is a missing component in wildlife habitat modeling

Commentary

Conspecific Attraction is a Missing Component in Wildlife Habitat Modeling

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ABSTRACT Wildlife biologists use knowledge about wildlife–habitat relationships to create habitat models to predict species occurrence across a landscape. Researchers attribute limitations in predictive ability of a habitat model to data deficiencies, missing parameters, error introduced by specifications of the statistical model, and natural variation. Few wildlife biologists, however, have incorporated intra- and interspecific interactions (e.g., conspecific attraction, competition, predator–prey relationships) to increase predictive accuracy of habitat models. Based on our literature review and preliminary data analysis, conspecific attraction can be a primary factor influencing habitat selection in wildlife. Conspecific attraction can lead to clustered distributions of wildlife within available habitat, reducing the predictive ability of habitat models based on vegetative and geographic parameters alone. We suggest wildlife biologists consider incorporating a parameter in habitat models for the clustered distribution of individuals within available habitat and investigate the mechanisms leading to clustered distributions of species, especially conspecific attraction. (JOURNAL OF WILDLIFE MANAGEMENT 72(1):331–336; 2008)

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KEY WORDS autologistic model, clustered distribution, conspecific attraction, habitat modeling, model error.

Wildlife biologists often consider habitat as a unifying concept in wildlife ecology (Morrison et al. 2006). The relationship between wildlife and the habitats they occupy has been one of the most intensively studied topics in the ecological literature (Manly et al. 2002, Morrison et al. 2006). Habitat modeling is commonly used in animal ecology to identify variables that describe a species' habitat, explain and predict species occurrence, determine available habitat for a species, and provide information for wildlife management (Mörtberg and Karlström 2005, Barry and Elith 2006). Wildlife biologists and naturalists have long described species' habitats based on vegetation and geographic characteristics (e.g., vegetation physiognomy and floristics, soil, slope, rainfall, climate; Grinnell 1917, Kendeigh 1945). These vegetative and geographic characteristics are commonly used in habitat models for predicting a species' habitat across a landscape.

The usefulness of a habitat model depends on its predictive ability. Errors in predictions are generally attributed to data deficiencies, missing parameters, error introduced by specifications of the statistical model, and natural variation (Elith et al. 2002, Barry and Elith 2006, Guisan et al. 2006). Few researchers, however, have incorporated the influence of intra- and interspecific interactions (e.g., conspecific attraction, competition, predator–prey relationships) in habitat models. Interactions among organisms are likely to influence the distribution of species within available habitat by causing organisms to separate or cluster. Processes leading to individuals separating from one another include interspecific niche separation (MacArthur 1958) and negative density dependence (Fretwell and Lucas 1970, Rosenzweig 1991). Processes leading to clustering include conspecific attraction (e.g., Stamps 1988) and predator–prey relationships (e.g., Penteriani et al. 2006).

There is considerable literature documenting the clustered distribution of organisms within identified habitat (Allee 1927, Darling 1952, Post 1974, Stamps 1988, Hays and Lidicker 2000) and investigating mechanisms driving clustered distributions (reviewed by Hildén 1965), including public information (Doligez et al. 1999, 2004*a*, *b*) and prospecting (Ward 2005). A primary mechanism driving clustered distributions is conspecific attraction. Researchers have found through field and laboratory experiments that conspecific attraction explains clustered distributions for some species (Stamps 1988, Muller 1998, Poysa et al. 1998, Ward and Schlossberg 2004; reviewed by Ahlering and Faaborg 2006).

Conspecific attraction, or the tendency of individuals to be attracted to and, thus, settle near individuals of their own species can be caused by a variety of underlying mechanisms. Researchers have suggested several reasons that individuals may be attracted to conspecifics when selecting habitat, including more potential mates, group vigilance, predator dilution, and use of conspecifics as indicators of habitat quality (e.g., resource type and quality, previously successful breeding; summarized by Muller et al. 1997). Thus conspecific attraction can potentially lead to clustered

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distributions within identified habitat (reviewed by Reed and Dobson 1993).

We have found that few wildlife biologists have considered conspecific attraction leading to the clustered distribution of a species when building habitat models. Thus, our goals were to 1) review primary approaches to habitat modeling, focusing on associated prediction errors, 2) discuss conspecific attraction as it pertains to habitat selection, 3) outline the importance of considering conspecific attraction leading to clustered distributions in habitat modeling studies, 4) review examples where species clustering was considered in habitat models, and 5) provide preliminary considerations for sampling designs to investigate conspecific attraction. We provide a case study using golden-cheeked warblers (*Dendroica chrysoparia*) to illustrate the potential influence of conspecific attraction on habitat models.

Habitat Modeling And Prediction Errors

Habitat models are also referred to as niche-based models, species-distribution models, and biogeographical models of species distributions (Araújo and Guisan 2006, Guisan et al. 2006, Segurado et al. 2006). The process of creating a habitat model includes several steps. Wildlife biologists collect data at numerous locations, noting a species' presence or absence and measuring biotic and abiotic characteristics at each location. Often, the biotic and abiotic characteristics measured are those that are either noted with ease or assumed to affect occupancy, such as vegetative and geographic characteristics. Wildlife biologists then use a statistical model to select the characteristics that are significantly correlated with species occupancy (Araújo and Guisan 2006). Next, wildlife biologists use the specified vegetative and geographic characteristics to generate a spatially explicit prediction of the areas where a species is most likely to occur across a landscape (Barry and Elith 2006).

Incorrect predictions of presence–absence in habitat models are generally attributed to 1) data deficiencies, 2) missing parameters, 3) error introduced by the specifications of the statistical model, and 4) natural variation (Elith et al. 2002, Barry and Elith 2006, Guisan et al. 2006). Guisan et al. (2006) categorized data deficiencies as measurement errors and systematic errors. Measurement errors include missing variables, small sample sizes, biased samples, biased estimates of presence or absence, and a paucity of presence or absence data (e.g., species of interest was ubiquitous in sampled area; thus, characteristics of unoccupied habitat were not obtained). Systematic errors include datum shifts and misclassifications of vegetative cover in remote sensing data. Missing parameters are biotic and abiotic factors that might directly or indirectly influence a species occurrence, but are not included in the habitat model. Errors introduced by specifications of the statistical model, also referred to as model-based error, occur when statistical models with inappropriate probability distributions are used to create the habitat model (Guisan et al. 2006). Natural variation is stochasticity in ecological processes.

Recent improvements in habitat modeling have focused on correcting data deficiencies and statistical model-based errors. MacKenzie and Royle (2005) provided study design suggestions to account for detection probabilities that are <1 and to decrease some measurement errors caused by inappropriate sample sizes, biased estimates of occurrence, and biased samples. The development of improved statistical models has provided wildlife biologists with better tools to predict occurrence (Legendre 1993, Guisan et al. 2002, Lehmann et al. 2002). Statistical modeling developments include methods for identifying spatial autocorrelation (Legendre 1993, Augustin et al. 1998, Lichstein et al. 2002), methods for including spatial structures into statistical models (Legendre 1993), and advancements in regression analysis provided by generalized linear models and generalized additive models (Guisan et al. 2002, Lehmann et al. 2002). The least discussed type of error in habitat model predictions is natural variation. Predicting species occurrence requires a thorough understanding of the spatial and temporal changes in ecological interactions between the target species and their environment, including intra- and interspecific interactions (Leathwick and Austin 2001, Huston 2002, Guisan et al. 2006). Guisan et al. (2006) called for integrating ecological interactions into modeling species' distributions.

Conspecific Attraction

The presence of conspecifics may positively or negatively influence habitat selection. Conspecific competition influences habitat selection (Svardson 1949) through individual fitness declines with increasing density of conspecific competitors (Fretwell and Lucas 1970, Rosenzweig 1991). However, selecting habitat based on the presence of conspecifics may increase individual fitness (e.g., Ward 2005). Donohue (2006) described 2 classes of advantages yielded by association with conspecifics: positive density-dependent or Allee effects (Allee 1927) and use of conspecifics as indicators of habitat quality. Effects of positive density dependence can be seen in a reduced risk of predation (Bertness and Grosholz 1985, Ray and Stoner 1994, Tyler 1995), increased foraging success (Clark and Mangel 1984), and increased access to mates (Allee 1927). The information provided by the presence of conspecifics, termed public information, can reflect characteristics of habitat quality that may not be easily detectable or accessible to new settlers, such as resource abundance, predator densities, and potential for reproductive success (Danchin et al. 2001; Doligez et al. 2004*a, b*; Donahue 2006). Black-legged kittiwakes (*Rissa tridactyla*; Danchin et al. 1998), yellow-headed blackbirds (*Xanthocephalus xanthocephalus*; Ward 2005), and collared-flycatchers (*Ficedula albicollis*; Doligez et al. 1999, 2004*b*) selected habitat based on the previous reproductive success of neighboring conspecifics.

Conspecific attraction has been documented in laboratory experiments for several taxa including coral reef fishes (Sweatman 1985), the porcelain crab (*Petrolisthes cinctipes*; Donahue 2006), the sanddollar (*Dendraster excentricus*; Highsmith 1982), and Panamanian grass anoles (*Anolis* *auratus*; Kiester 1979). Conspecific attraction also has been observed in manipulative and mensurative field experiments for several taxa. Nest-box selection by naïve house wrens (*Troglodytes aedon*; i.e., individuals entering their first breeding season) was correlated with the presence of conspecifics (Muller et al. 1997). Muller (1998) found that habitat selection by a species of grasshopper (*Ligurotettix coquilletti*) was influenced by the presence of conspecifics. Similarly, conspecific attraction was found to influence habitat selection for mallards (*Anas platyrhynchos*; Poysa et al. 1998), pied flycatchers (*Ficedula hypoleuca*; Alatalo et al. 1982), black-capped chickadees (*Peocile atricapillus*; Ramsay et al. 1999), and black-capped vireos (*Vireo atricapilla*; Ward and Schlossberg 2004). In some cases, individuals selected areas described as suboptimal habitat due to the influence of conspecific attraction (Tiainen et al. 1983). Conspecific attraction appears to influence habitat selection for a variety of species, which may cause spatial clustering of individuals.

Why Is Conspecific Attraction Important For Habitat Modeling?

Frequently, habitat models predict that a species will be present in certain areas, but some of those areas are not occupied. Likewise, species are found in some areas where the habitat model does not predict presence. These limitations in the predictive ability of habitat models are often attributed to missing habitat variables (Barry and Elith 2006). However, conspecific attraction leading to clustering of individuals within predicted habitat may explain the limitation of some habitat models (see Lichstein et al. 2002). Also, conspecific attraction may explain the presence of individuals outside of predicted habitat because individuals were attracted to the presence of conspecifics in adjacent areas of occupied, predicted habitat. Incorporating a parameter for clustering of individuals in habitat modeling can improve model predictions (Augustin et al. 1996, Lichstein et al. 2002).

Improving habitat models by incorporating a parameter for clustered distributions may provide more accurate predictions of a species' distribution and can assist wildlife managers by enabling them to focus on areas where occupancy is more likely. Improving a habitat model by incorporating such a parameter may reduce the search for missing vegetative and geographic parameters (Legendre 1993). Greater emphasis can then be placed on management, conservation, and investigating the mechanisms driving the clustered distribution of conspecifics within identified habitat, especially conspecific attraction.

Habitat Modeling For Clustered Distributions

Our review of the literature indicated that few wildlife biologists have considered clustered distributions of conspecifics in habitat models. However, statisticians and some wildlife biologists have used several statistical methods to address clustered distributions of conspecifics, beginning with agricultural pests and bacteria (Neyman 1939) and plants (Clark and Evans 1954). Logistic regression is commonly used to develop habitat models because wildlife survey data are typically treated as presence–absence (i.e., binomial) data.

We found 3 examples in the literature of statistical approaches to incorporate clustered distributions in habitat modeling. Smith (1994) used logistic regression with additional variables to explain and predict the spatial distribution of mountain sorrel (*Oxyria digyna*) in Britain. Smith (1994) assigned orders to adjacent cells to indicate their degree of proximity to a focal cell in a gridded sampling design and used the number of detections in each order of adjacent cells to generate the new variables. This method addressed autocorrelation in species distribution that could not be addressed by fitting other autocorrelated environmental variables associated with occurrence. Smith (1994) found that incorporating the additional variables improved both the explanatory and predictive capabilities of the habitat model. Augustin et al. (1996, 1998) created a habitat model to predict red deer (*Cervus elaphus*) distribution in Scotland using a gridded sampling design. Augustin et al. (1996, 1998) analyzed survey data by extending a logistic regression model to incorporate a parameter indicating presence–absence at neighboring survey locations to account for autocorrelation in red deer distributions. Augustin et al. (1996) found that including this parameter significantly improved their estimate of the spatial distribution of red deer across the landscape compared to a logistic-regression model without the added parameter. Cornulier and Bretagnolle (2006) used point-process statistics to test for the clustered distribution of nests with respect to available habitat for little owl (*Athene noctua*) and Montagu's harrier (*Circus pygargus*) in France. Cornulier and Bretagnolle (2006) suggested using point-process statistics (the $K(r)$ function [see Ripley 1977]) to analyze data consisting of point locations within a defined area to detect interactions in the point pattern such as aggregation or inhibition. Cornulier and Brentagnolle (2006) found that observing aggregation depended on the spatial scale considered and on accounting for available habitat for each species.

These examples demonstrate several methods of incorporating metrics that reflect clustered distributions, and that such approaches can improve the predictive ability of habitat models. These habitat modeling methods seem especially applicable to management and conservation of wildlife because they extrapolate sample data to predict a species' distribution across a landscape.

Sampling For Conspecific Attraction

To our knowledge, few field experiments have used sampling designs to explicitly evaluate the occurrence of conspecific attraction across large spatial scales and its influence on a species' distribution. Wildlife biologists have used manipulative experiments in the lab (e.g., Kiester 1979, Sweatman 1985, Donahue 2006) and field (Muller 1998, Ward and Schlossberg 2004, Ahlering and Faaborg 2006) to determine if species respond to conspecific cues (e.g., presence of conspecifics, visual models, vocalizations). Also, research has focused on correlating territory or nest-site selection with the presence of conspecifics on relatively small spatial scales (e.g., Tiainen et al. 1983, Muller et al. 1997, Doligez et al. 1999). Conspecific attraction is likely to influence a species' distribution on both small and large spatial scales; thus, sampling for conspecific attraction should be incorporated into occupancy studies.

Estimates of abundance or occupancy are usually based on designs where sampling is conducted on an annual basis (e.g., winter surveys for ungulates). Thus, these estimates are influenced primarily by sampling variation (e.g., variability among counts, variability among plots). Most studies focus little effort on quantifying process variation, primarily based on the assumption that organisms at the plot level are affected equally by changes in environment or demographic process. Sampling designs for conspecific attraction must account for temporal and spatial variation because these factors are inherent to conspecific attraction. Depending on the reproductive phenology of the study species, conspecific attraction may occur in one week or occur irregularly across a season (temporal variation). Also, conspecific attraction may occur on small or large spatial scales both within and among habitat patches (spatial variation).

Many sampling designs are potentially useful for evaluating conspecific attraction in wildlife species, including multistage sampling (Thompson 2002), ranked-set sampling (Thompson et al. 1998), double-sampling (Bart and Earnst 2002), or adaptive sampling (Thompson and Seber 1996, Thompson 2002). Sampling designs developed for clustered populations are focused primarily on incorporating clustering into survey plots, often with the benefit of reduced variance and increasing estimator precision (Thompson and Seber 1996, Thompson 2002). Plot delineation for evaluation of conspecific attraction is difficult, especially in the case of breeding birds. Vegetative and geographic metrics are frequently used to determine plot size, shape, and placement (Morrison et al. 2001). However, multiple patches of high-quality habitat may exist within a woodland, wetland, or meadow. Development and application of designs to differentiate between clustering due to conspecific attraction and clustering due to the clustering of other habitat characteristics (e.g., resources, vegetation structure) should become an active area of research in habitat modeling.

A Case For Conspecific Attraction In Golden-Cheeked Warblers

Wildlife biologists have described golden-cheeked warbler habitat as mature oak–juniper (*Quercus–Juniperus*) woodland (Pulich 1976, Kroll 1980, Ladd and Gass 1999, Magness et al. 2006). Researchers have used habitat models to predict golden-cheeked warbler occupancy using parameters associated with mature oak–juniper woodland (Cummins 2006, DeBoer and Diamond 2006, Jones 2006). Prediction errors of the habitat models were attributed to data deficiencies (e.g., small sample size, inability to detect fine distinctions in habitat parameters) and missing habitat parameters (e.g., woodland stand age, habitat structure, regional variability in warbler habitat; DeBoer and Diamond 2006) or the errors were not addressed (Cummins 2006, Jones 2006). Conspecific attraction was not considered in these habitat models for golden-cheeked warblers.

Based on the previous research noted above, we created a habitat model using a Landsat map of east-central Texas, USA, prepared by the University of Texas at Austin Center for Space Research. University of Texas at Austin Center for Space Research used sub-pixel classification to identify the vegetation species in each pixel of the Landsat map. We used an ArcGIS™ Spatial Analyst Neighborhood Statistics procedure with a 400-m radius moving window on the classified map to identify areas with <40%, 40–60%, and >60% oak–juniper woodland composition. We based these composition classes on research showing that golden-cheeked warblers were increasingly likely to occupy areas with greater percentage oak–juniper woodland composition (Cummins 2006, DeBoer and Diamond 2006) and were unlikely to occupy areas with oak–juniper woodland composition <40% (Magness et al. 2006).

We conducted 6 point-count surveys from 1 April to 1 June 2006 at 211 survey stations. We used these data to evaluate the predictive ability of our habitat model for predicting golden-cheeked warbler occurrence. We detected golden-cheeked warblers at 20% of point-count stations in <40% oak–juniper woodland ($n = 127$), 62% of stations in 40–60% ($n = 53$), and 84% of stations in >60% ($n = 31$). We detected warblers in some areas where the habitat model did not predict occupancy (<40% oak–juniper woodland) and did not in some areas where the habitat model predicted occupancy (>60% oak–juniper woodland).

We examined our data for evidence of conspecific attraction to potentially explain errors in the predictive ability of the habitat model. Our objective was to determine if a warbler detection in a higher oak–juniper composition class would increase the probability of a detection in a neighboring, lower oak–juniper composition class. For each point-count station where we detected a warbler, we calculated the proportion of point-count stations both within 400 m and in lower percent oak–juniper composition where we also detected warblers. For each point-count station where we did not detect a warbler we calculated the proportion of point-count stations both within 400 m and in lower percentage oak–juniper composition where we detected warblers.

Forty-one point-count stations met our criteria having neighboring points both within 400 m and in lower oak–juniper composition classes. For stations where we detected a warbler, the proportion of adjacent stations in a lower composition class that were occupied was 0.64 (SE = 0.01, median = 0.50, mode = 1, $n = 27$). For stations where we did not detect a warbler, the proportion of adjacent stations in a lower composition class that were occupied was 0.48 (SE = 0.03, median = 0.24, mode = 0, $n = 14$). The difference between these proportions was 0.16 (95% CI = –0.16–0.48). Although this interval was not statistically different, it contained differences we suggest may be biologically important for understanding the distribution of golden-cheeked warblers.

Our preliminary analysis found that warblers were more likely to be present in a lower composition class if there was a warbler present in an adjacent higher composition class and we suspect this may be due to conspecific attraction. Our findings suggest that the presence or absence of conspecifics in adjacent areas influences the distribution of golden-cheeked warblers within mature oak–juniper woodland. We collected our occupancy data as part of a larger monitoring project that was not explicitly designed to investigate conspecific attraction or clustered distributions and may need to be adjusted to more rigorously pursue this objective. Future research is needed to determine if golden-cheeked warblers are clustered within available habitat. If clustered distributions are observed, future research is needed to determine if conspecific attraction is a mechanism driving clustered distributions.

Management Implications

We urge wildlife biologists to consider conspecific attraction and the resulting clustered distributions in their habitat models to explain and predict species occurrence across a landscape. We expect that habitat models incorporating a parameter for clustered distributions will have improved predictive ability and, thus, become more useful tools for management and conservation efforts.

Information on conspecific attraction leading to clustering can influence management needs and objectives for wildlife species. The management objective may be to conserve or restore all potentially suitable habitat for a species of interest. In this case, we recommend creating habitat models with and without a parameter for clustered distributions. The habitat model without a clustering parameter is likely to identify all areas with the specified vegetative and geographic characteristics thought to be suitable for the species. The habitat model with a parameter for clustered distributions can be used to determine if the species of interest is indeed clustered within areas identified by the habitat model constructed with vegetative and geographic parameters only. If clustering is observed, the absence of the species in areas predicted to be suitable based on vegetative and geographic parameters may not be due to unsuitability of these characteristics, but to inter- and intraspecific interactions, especially conspecific attraction. This information on clustered distributions can be used to support management actions to conserve and restore currently unoccupied habitat. Especially in the case of endangered species, there may not be enough individuals to occupy all potential habitat, and to recover a species, potential habitat that is currently unoccupied will need to be occupied in the future.

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