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Editors

Vertebrate Conservation and Biodiversity

Springer

Vertebrate Conservation and Biodiversity

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Vertebrate Conservation and Biodiversity

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and
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Introduction

Vertebrate Conservation and Biodiversity

This book brings together a selection of original studies submitted to *Biodiversity and Conservation* addressing aspects of the conservation of vertebrates and biodiversity, especially those in terrestrial habitats[1]. Vertebrates are, along with plants, the best-known and most intensively studied components of biological diversity on Earth. As such, studies on vertebrates may be expected to provide models for other groups, but at the same time they pose particular problems because of their relative mobility, which can be transcontinental in migratory birds. In addition, many mammals and fish are also subject to extensive human exploitation for food or sport.

Two contributions consider the issue of habitat modification in relation to birds, mammals and also reptiles, one carried out in East African coastal forests, and the other a planted tree crop in Nicaragua; in both cases primary and relict forests were critical to the survival of the most specialized species.

The articles concerned with mammals cover the conflict caused by large carnivores attacking livestock in Africa, mammals in ranches and other habitats in the Brazilian Cerrado, mammals in the Caxiuanã National Forest in Brazil (monitored using a camera phototrapping programme), koalas and a wombat in Australia, the maned wolf in the Brazilian Cerrado and Pantanal, bats in Madagascar, marmosets in Brazil, and endemic primates in Kenya. Problems of surveying carnivores at different scales are revealed by the comparison of different methods in the Mediterranean.

The contributions on birds included here fall into two main categories: the relationships of the avifauna to habitat types and changes, and studies on particular species. In the former category are studies comparing abundance and diversity in grassland and non-grassland habitats in Hungary, birds in wetland fragments in the neotropics, responses to prescribed burning in prairies in the USA, relationships to human activities in the Córdoba Mountains of Argentina, and ones in montane forests of different degrees of modification in Ecuador. The studies on particular species presented encompass ones on the Lower Grey Shrike in France and Spain, the migratory Lesser Grey Shrike in Europe, Great Spotted Woodpeckers in Switzerland, the Sooty Shearwater on the South American coast, the Sierra Madre Sparrow in México, and Blue-winged Macaws in Brazil. Also included is a critical study of bird endemism and biodiversity in the different subregions of China which may help direct conservation efforts in the country.

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[1] A series of contributions including ones on fish was included in a previous volume of collected papers in this series, *Marine, Freshwater, and Wetlands Biodiversity Conservation* [Topics in Biodiversity and Conservation Vol. 4], edited by D. L. Hawksworth & A. T. Bull (Springer, Dordrecht, 2006; 399 pp., ISBN 10: 1 4020 5733 4, 13: 978 1 4020 5633).

   ⓐ Springer

Reptiles and amphibians include many of the most endangered vertebrates, with anuran decline being an issue of ongoing major world concern. The situation with anurans is exemplified here by a study of 131 species in the Brazilian Cerrado leading to the design of a network of conservation areas. Other investigations reported on these groups here include ones on amphibian diversity hot-spots in the Tanzanian mountains, characterization of factors favouring these organisms in a National Park in Portugal, and the decline of Loggerhead Turtles on a beach in Turkey. The issue of exploitation is also represented by reports on the reptile skin trade in México, and the sale of snakes in markets and stores in Brazil as pets, sources of traditional medicines, or uses in religious rites. Another aspect is the effect of introduced species on others, exemplified here by the threat posed to native species by the Boa constrictor on a Mexican island.

This series of studies is presented together here to provide an indication of current research activities, that will be of value to students undertaking courses in aspects of biodiversity and conservation. They can be viewed as a series of case studies that will expose students to primary research being conducted now. As such they will complement the necessarily less-detailed specific information in textbooks and secondary review articles.

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Responses of plant and bird communities to prescribed burning in tallgrass prairies

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Abstract Historic losses and fragmentation of tallgrass prairie habitat to agriculture and urban development have led to declines in diversity and abundance of plants and birds associated with such habitat. Prescribed burning is a management strategy that has potential for restoring and rejuvenating prairies in fragmented landscapes, and through such restoration, might create habitat for birds dependent upon prairies. To provide improved data for management decision-making regarding the use of prescribed fire in tallgrass prairies, we compared responses of plant and bird communities on five burned and five unburned tallgrass prairie fragments at the DeSoto National Wildlife Refuge, Iowa, USA, from 1995 to 1997. Overall species richness and diversity were unaffected by burning, but individual species of plants and birds were affected by year-treatment interactions, including northern bobwhite (*Colinus virginianus*) and ring-necked pheasant (*Phasianus colchicus*), which showed time-delayed increases in density on burned sites. Analyses of species/area relationships indicated that, collectively, many small sites did make significant contributions to plant biodiversity at landscape levels, supporting the overall conservation value of prairie fragments. In contrast, most birds species were present on larger sites. Thus, higher biodiversity in bird communities which contain area-sensitive species might require larger sites able to support larger, more stable populations, greater habitat heterogeneity, and greater opportunity for niche separation.

Keywords DeSoto National Wildlife Refuge · Grassland birds · Grassland plants · Prairie restoration · Prescribed burning · Tallgrass prairie

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Introduction

The structure and function of grasslands worldwide have been disrupted by habitat fragmentation, the removal of native grazers and altered fire frequency (Samson and Knopf 1994). Such changes threaten the integrity of native grassland communities of plants and animals in many ways and at many levels (Janzen 1983; Collins 2000; Cully and Michaels 2000). Habitat fragmentation alters size, spacing and context of habitat patches, which can result in an increase in the local rate of extinction of plant and animal species by reducing fecundity, population size and colonization of species from similar habitats (Noss and Csuti 1992). Compounding the effects of fragmentation, extirpation of fire and loss of populations of native grazers can lead to encroachment of woody vegetation and loss of grassland (Leach and Givnish 1996; Collins et al. 1998).

In addition to effects on plant communities, the use of prescribed fire to enhance grasslands is of increasing relevance as regional losses, isolation and fragmentation of grasslands have led to corresponding declines in local population densities of many species of grassland-dependent birds (Askins 1993; Vickery et al. 1994). Area-sensitive species in grasslands are especially vulnerable to these effects (Herkert 1994a, b) which can create specific physical, environmental and ecological changes (Yahner 1988) that lower habitat quality for area-sensitive species that have been historical specialists of grassland habitats. Prescribed burning cannot change the size, shape, or area of a site, but it can alter vegetation density, vegetation structure and habitat heterogeneity. Thus, prescribed burning can be a management tool for providing appropriate habitat to vegetation-sensitive species.

With these concerns in mind, we examined the effects of prescribed burning on plant and bird communities in tallgrass prairie fragments in order to determine (1) changes in the structure of vegetation and abundance of plant species following burning in tallgrass prairie habitats; (2) the response of individual species of resident grassland-dependent breeding birds; and (3) the effects of prescribed burning on overall plant and avian community species richness and diversity. The questions we wished to answer were (1) would differences in management on fragmented prairie habitat lead to different communities of plants and birds associated with such habitat and (2) would differences in management treatments be effective in moving communities of low species diversity toward higher levels of biodiversity. Our null hypothesis was that there would be no differences in characteristics of these communities on burned versus unburned sites.

Study area

The DeSoto National Wildlife Refuge (DNWR) was established along the Missouri River in Iowa and Nebraska (USA) in 1959 to conserve wetlands used as resting and staging areas for migrating waterfowl. In addition to wetlands (approximately 1000 ha), DNWR's most abundant habitats are forests (1350 ha) and native tallgrass prairie (665 ha). As a result of settlement and conversion of prairie communities in this region to agriculture and pasture lands beginning in the second half of the 19th century, no large blocks of native tallgrass prairie vegetation remained by the time DNWR was created. Given the rarity of tallgrass prairie in the upper Midwest, prairies at DNWR was re-established intentionally on marginal croplands within

refuge borders to conserve this historic native habitat and its biological diversity. Tallgrass prairie exists today at DNWR in fragments of 3–10 ha that are actively managed by prescribed burning every third year.

Initial conversions of croplands into prairie were made on smaller sites, sites with configurations that made cultivation difficult, sites with historically poor crop yields, sites with potentially high wildlife value, or sites with a combination of these traits. Restoration efforts began with a single site in 1965 that was planted with one species, big bluestem (*Andropogon gerardii*), a C_4 grass often considered an indicator of tallgrass prairies. Six additional sites were converted in 1972–1974 with the planting of 3–5 species of warm season C_4 grasses but no forb species. In 1980–1983, five more sites were converted with plantings of six species of grasses and up to five species of forbs. Thus, all early restoration efforts were characterized by low species diversity (1–11 species). Management following plantings was site-specific according to site characteristics and management objectives, but typically involved burning and mowing at 3–4 year intervals.

By the 1990s, vegetation in these tallgrass prairie fragments at DNWR was dominated by big bluestem, Indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*) and various species of goldenrod (mainly *Solidago canadensis*). The most abundant bird species were common yellowthroats (*Geothlypis trichas*) and field sparrows (*Spizella pusilla*). The general status of sites had been monitored since restoration, but quantitative measurements of plant and animal community composition were limited. Our study was the first comprehensive quantitative examination of such restored sites at DNWR and the first direct comparison of plant and bird communities on burned and unburned sites there.

Materials and methods

Sampling of plant and bird communities

In January 1995, 10 independent noncontiguous sites of native tallgrass prairie at DNWR were selected for evaluation as statistical sampling units. All sites possessed similar physical characteristics and site histories, and all were within a relatively narrow size range (3.0–9.3 ha) typical of prairie fragments on the refuge. Five sites were designated randomly for burning while the other five were left untreated. Burning was completed on all sites between 22 April and 11 May in 1995. Fires were ignited at multiple points along a prescribed line using kerosene drip torches and completed in 1–2 h. Each site retained its treatment designation for sampling through 1997, but burned sites were burned only in 1995 and were not reburned in subsequent years, an approach that permitted us to track changes on these sites subsequent to burning.

Regeneration on burned sites was rapid and vegetation was re- established by the time sampling began in June. Vegetation in each prairie fragment was examined using five 50×50 m macroplots sufficient to cover all parts of the site. On all sites, macroplots were nonoverlapping and covered the entire prairie fragment so that all parts of the site were sampled and total diversity of the site's plant community could be assessed. Six 50-m belt transects were established perpendicular to the baseline of each macroplot at random intervals. Plant species' composition and percent ground cover were determined in five 25.4×50.8 cm microplots placed randomly along

each belt (30 microplots/macroplot). Estimates of cover were used as the primary index of plant abundance and were made using a visually calibrated frame. Presence of individual species was used to determine species richness. Species diversity was determined by using the Shannon Index (H') (Shannon and Weaver 1949),

$$H' = -\sum_i (p_i \ln p_i),$$

where p_i is the proportion of individuals of the ith species. Species dominance on each site was determined using the Simpson index (C) (Pielou 1975), which measures dominance as the probability that two randomly selected individuals from a community will belong to the same species and is calculated as

$$C = \sum_i p_i^2.$$

From 20 May through 10 July 1995–1997, the density and diversity of resident breeding bird species was determined by a census of singing males by using the spot mapping method (International Bird Census Committee 1970). Each site was visited 10–12 times each year. Birds were designated as grassland residents if they established home ranges that were confined to the site or, if extending beyond the site, remained within adjacent grassland habitat. We also included two nonterritorial species, brown-headed cowbird (*Molothrus ater*) and mourning dove (*Zenaida macroura*), as members of the resident grassland community, as well as American goldfinch (*Carduelis tristis*), a species that typically established territories subsequent to our sampling period. Because these species invested most of their time in grassland fragments, we included them in the resident bird community. In these species, we censused males within the grassland whether vocalizing or not. Composite results of all visits were used to determine the abundance of individual species, species richness and diversity.

Numerical and statistical analyses

We compared community species richness, diversity, dominance and species cover (plants) and density (resident birds) through a repeated measures analysis of variance, with years as repeated measures, to determine whether these variables were independent of treatment. Plant cover estimates were arcsine-transformed before analysis. $P \leq 0.05$ was considered significant. For species that had significant treatment–year interactions, we conducted unpaired t-tests to determine in which years treatment affected species abundance. Bonferroni corrections were applied to account for the multiple years being compared. To determine the cumulative effect of all sites on the species richness of both plant and bird communities, we performed a rarefaction analysis (Krebs 1989) that provided an estimate of the expected number of species present per sampled area.

We measured similarity of plant communities between plots of different treatments and years using the Jaccard index, a measure of association (similarity) between communities based on differences in species presence and absence. The Jaccard Index provides an easily interpreted measure of association between communities but should not be analyzed using standard analysis of variance

(Dyer 1978). To extend analysis of the Jaccard Index across multiple factors, we analyzed species composition of plant communities between treatments and among years using the linear model proposed by Dyer (1978). The baseline similarity estimated by the model is a measure of the average Jaccard similarity between sites in the same treatment and year. The coefficient estimated for the effect of treatment provides a measure of change in similarity between treatments in the same year. The coefficient estimated for the effect of year provides a measure of change in similarity on the same site in different years. Based on a permutation test applied separately to treatment and year factors (Edgington 1995), we evaluated whether communities from different treatments or years were dissimilar to one another. The impact of each species on the Jaccard computations also was estimated.

Results

Species richness, diversity and dominance of tallgrass plant and bird communities were independent of treatment (Tables 1, 2). Overall, species richness and diversity in plant and bird communities were low on all sites. Year effects were significant influences on species richness in both plant and bird communities (Table 1) and affected species diversity and dominance in bird communities (Table 2). Resident bird communities were lower in richness and diversity and higher in dominance in 1995 compared to subsequent years. Proportional treatment differences in species richness were greatest immediately after burning (1995) in both plants and birds, but diminished in subsequent years.

Plant communities at DNWR contained few species on individual sites (range 8–24). Forty-six percent of ground cover was contributed by one species of grass, *A. gerardii*, and one species of forb, *S. canadensis*. Rarefaction analysis revealed that the entire array of sites collectively contributed many more species than any individual site, with the upper asymptote of expected number of species in any given year (20–30 species) not being approached until sampling area exceeded 30 ha (Fig. 1), more than three times the area of the largest individual site.

One plant species (*Cassia fasciculata*) showed sensitivity to treatment effects, five (*A. gerardii, C. fasciculata, P. virgatum, Schizachyrium scoparium* and *S. nutans*) to year effects (Table 3) and three (*C. fasciculata, Polygonum pensylvanicum* and *S. scoparium*) to treatment–year interactions ($P = < 0.01, 0.03, 0.02$, respectively).

Table 1 Effect of management treatment and year on species richness in communities of plants and resident birds in tallgrass prairie fragments at DNWR, Iowa, USA, 1995–1997

Community	1995			1996			1997			P (Treatment)	P (Year)
	Mean	SE	Range	Mean	SE	Range	Mean	SE	Range		
Plants	14.4	1.6	8–22	21.2	1.2	14–26	19.6	1.5	13–25	0.63	< 0.01
Burned	16.2	2.2	11–22	20.6	2.0	14–24	20.2	2.3	13–25	–	–
Untreated	12.6	2.2	8–19	21.8	1.5	18–26	19.0	2.1	14–25	–	–
Resident Birds	7.8	1.0	3–14	10.2	0.4	8–12	10.4	0.7	7–14	0.09	< 0.01
Burned	9.0	1.6	4–14	10.6	0.5	9–12	11.8	0.8	10–14	–	–
Untreated	6.6	1.0	3–8	9.8	0.5	8–11	9.0	0.7	7–11	–	–

Means of species richness compared through repeated measures (year) analysis of variance

Table 2 Effect of management treatment and year on species diversity (Shannon index) and dominance (Simpson index) of plant and resident grassland bird communities in burned and untreated tallgrass prairies at DNWR, Iowa, USA, 1995–1997

| Community | 1995 | | | | 1996 | | | | 1997 | | | | P (Treatment) | P (Year) |
| | Burned | | Untreated | | Burned | | Untreated | | Burned | | Untreated | | | |
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE		
Plants														
Diversity	1.87	0.13	1.82	0.12	1.95	0.09	1.92	0.07	1.85	0.12	1.80	0.13	0.75	0.24
Dominance	0.22	0.03	0.21	0.03	0.21	0.02	0.23	0.02	0.24	0.03	0.26	0.05	0.79	0.19
Resident birds														
Diversity	1.57	0.13	1.27	0.12	1.74	0.07	1.72	0.17	1.91	0.06	1.57	0.09	0.11	< 0.01
Dominance	0.27	0.02	0.35	0.03	0.26	0.03	0.26	0.05	0.22	0.03	0.28	0.02	0.23	0.04

Means of species diversity and dominance compared through repeated measures (year) analysis of variance

C. fasciculata, a N-fixing legume, grew only on burned sites and only 1–2 years after burning occurred. *A. gerardii* declined in cover on burned and unburned sites with time. *S. scoparium* and *S. nutans* declined in coverage on untreated sites over time. Among species that demonstrated treatment–year interactive effects, *C. fasciculata* had greater coverage on burned sites in 1996 ($t_8 = -3.4$, $P = 0.03$, Bonferroni corrected). Other treatment differences in individual years among species were not significant ($P \geq 0.43$, *d.f.* = 8, all cases, Bonferroni corrected). In community structure, graminoid, shrub and total plant cover were affected by year but not treatment (Table 3). Graminoid and total plant cover declined, and shrub cover increased, with time on burned and unburned sites. Heights of individual species were not directly measured, but we observed visually similar plant height and stratification on burned and unburned sites, supporting the inference from coverage data that community structure was similar in both treatments.
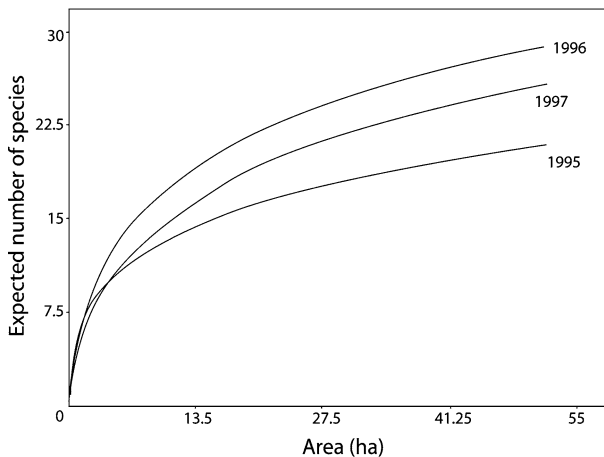
Fig. 1 Rarefaction analysis estimating the expected number of plant species per sampled area in tallgrass prairies at DNWR, Iowa, USA, 1995–1997. Different years plotted separately

Table 3 Percent ground coverage of 10 most common plants in burned and untreated tallgrass prairies at DNWR, Iowa, USA, 1995–1997

Community	1995				1996				1997				P (Treatment)	P (Year)
	Burned		Untreated		Burned		Untreated		Burned		Untreated			
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE		
Andropogon gerardii	11.0	4.5	9.9	2.0	9.8	4.3	7.7	2.6	5.1	1.1	3.8	1.1	0.75	< 0.01
Asclepias spp.	1.2	0.5	0.8	0.5	2.2	0.5	1.0	0.6	0.9	0.5	0.2	0.1	0.10	0.07
Bouteloua curtipendula	1.1	1.1	3.8	2.2	4.6	3.2	4.7	3.5	4.0	2.4	3.2	2.2	0.74	0.42
Carex spp.	0.3	0.1	0.3	0.3	0.2	0.1	0.5	0.3	0.1	< 0.1	0.9	0.7	0.48	0.75
Cassia fasciculata	0.0	0.0	0.0	0.0	3.1	1.2	0.0	0.0	0.7	0.4	0.0	0.0	0.01	< 0.01
Panicum virgatum	5.1	1.6	9.6	2.7	5.1	1.5	7.5	2.2	3.4	1.0	5.5	1.3	0.20	< 0.01
Polygonum pensylvanicum	0.9	0.7	0.7	0.5	0.8	0.7	1.7	1.3	0.4	0.3	2.2	1.6	0.56	0.70
Schizachyrium scoparium	2.9	1.9	5.1	0.9	1.3	0.8	2.3	0.9	1.8	0.9	1.1	0.8	0.43	0.01
Solidago canadensis	16.2	5.4	10.9	5.1	12.7	4.5	10.5	3.4	13.0	4.2	15.5	4.9	0.88	0.33
Sorghastrum nutans	6.7	2.2	5.3	1.6	3.6	1.0	1.6	0.4	4.5	1.9	1.5	0.6	0.28	< 0.01
Total grass	29.7	3.7	35.0	3.4	27.2	5.1	25.4	4.0	20.0	2.7	17.0	2.4	0.99	< 0.01
Total forb	25.5	7.4	15.3	4.3	22.6	6.3	16.0	4.3	17.6	5.3	20.8	5.9	0.66	0.80
Total shrub	0.0	0.0	0.0	0.0	1.5	0.9	1.0	0.3	0.7	0.4	0.6	0.4	0.77	< 0.01
Total plant cover	55.2	5.0	50.3	4.8	51.4	6.1	42.4	2.5	38.4	5.1	38.3	4.1	0.38	0.01

Means of cover of different treatments compared through repeated measures (year) analysis of variance. Percent cover arcsine-transformed for analysis

Plant communities on individual sites were dissimilar in year-to-year comparisons but not in comparisons between sites of different treatments. The average similarity of plots within the same treatment and year (baseline coefficient) was 0.43 (SE = 0.01). Burned and unburned sites contained essentially the same species with little decrease in similarity between treatments (treatment coefficient = − 0.009, SE = 0.007, $P = 0.45$), but site similarity decreased over time (year coefficient = − 0.086, SE = 0.008, $P < 0.01$). More than 70% of dissimilarity between sites was contributed by just 10 species (Table 4).

Burning had no effect on the total density of resident breeding birds or on the densities of individual species (Table 5). In contrast to plant communities, most species of birds were present on most sites. Fourteen species established territories on at least one site in every year, a total only slightly higher than the average number of species found on untreated sites (9–10) and burned sites (10–12) in 1996 and 1997. In 1996, the year of highest resident bird density, rarefaction analysis revealed an upper asymptote of 12 species at 10 ha (Fig. 2), a species–area relationship approximated on most of the larger sites. In years of lower population densities, upper asymptotes of expected numbers of species were associated with much larger areas (30–40 ha).

In population density, treatment effects were not statistically significant in any species. It is noteworthy, however, that field sparrow averaged twice the density on untreated sites as on burned sites throughout the study (Table 5). This difference was marginally nonsignificant ($F_1 = 4.6$, $P = 0.06$). However, our small sample size, combined with consistently large differences in density on different treatments in every year, makes it inappropriate for us to assert a conclusion of ''no effect'' of prescribed burning in this species. Rather, it raises the possibility of a biologically important effect that was not detected due to low statistical power associated with the test.

Total bird density and densities of seven species were affected by year, suggesting that annual populations of resident breeding birds on these sites varied significantly. In affected species, densities were lowest in 1995 and, in five of these, highest in 1996, as was total density. Six species showed the greatest proportional treatment differences in 1995, immediately after burning. One songbird, eastern kingbird (*Tyrannus tyrannus*) ($F_2 = 4.9$, $P = 0.02$), and two upland game birds, northern bobwhite

Table 4 Relative contribution to dissimilarity index of 10 plant species in plots sampled at DNWR, Iowa, USA, 1995–1997

Species	(%)
Graminoids	
Andropogon gerardii	4.76
Bouteloua curtipendula	4.29
Carex spp.	5.79
Panicum virgatum	10.91
Schizachyrium scoparium	8.15
Sorghastrum nutans	10.91
Forbs	
Apocynum cannabinum	3.84
Asclepias spp.	9.48
Polygonum pensylvanicum	3.41
Solidago canadensis	10.91
Total	72.45

Table 5 Densities (males/10 ha) of 12 most common resident grassland bird species in burned and untreated tallgrass prairies at DNWR, Iowa, USA, 1995–1997

| Community | 1995 | | | | 1996 | | | | 1997 | | | | P (Treatment) | P (Year) |
| | Burned | | Untreated | | Burned | | Untreated | | Burned | | Untreated | | | |
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE		
American Goldfinch	0.4	0.3	< 0.1	< 0.1	2.1	0.7	1.9	0.5	2.3	0.9	3.1	0.8	0.92	< 0.01
Brown-headed Cowbird	0.1	0.1	0.0	0.0	1.7	0.5	2.4	1.0	1.4	0.4	1.5	0.3	0.60	< 0.01
Common Yellowthroat	2.1	0.9	4.9	1.0	16.4	4.0	17.0	7.3	9.6	2.5	7.2	1.7	0.92	< 0.01
Dickcissel	4.3	1.3	1.9	1.2	3.2	1.1	2.6	0.8	1.9	0.3	2.0	1.2	0.44	0.27
Eastern Kingbird	0.2	0.1	0.1	0.1	0.9	0.2	2.3	0.6	0.8	0.3	0.6	0.2	0.29	< 0.01
Field Sparrow	2.5	0.8	5.8	1.3	5.0	0.9	10.1	2.6	3.8	0.6	7.0	2.2	0.06	0.03
Grasshopper Sparrow	0.2	0.2	0.2	0.2	0.2	0.1	0.0	0.0	0.3	0.2	< 0.1	< 0.1	0.24	0.66
Northern Bobwhite	0.2	0.1	0.2	0.1	0.8	0.3	0.7	0.3	1.4	0.3	0.4	0.1	0.14	0.01
Red-winged Blackbird	1.6	0.8	0.9	0.6	2.4	0.9	2.4	2.0	1.2	0.7	1.0	1.0	0.82	0.09
Ring-necked Pheasant	0.3	0.1	0.2	0.1	1.7	0.5	1.7	0.4	2.0	0.6	0.2	0.1	0.21	< 0.01
Vesper Sparrow	0.3	0.3	0.3	0.3	0.2	0.2	1.7	1.0	0.3	0.1	0.1	< 0.1	0.37	0.14
Western Meadowlark	0.1	0.1	< 0.1	< 0.1	0.5	0.5	0.1	0.1	0.2	0.2	0.0	0.0	0.41	0.29
Total Density	13.0	2.5	15.0	1.7	35.6	6.6	43.0	8.2	25.8	3.9	23.6	3.6	0.68	< 0.01

Means of species densities of different treatments compared through repeated measures (year) analysis of variance
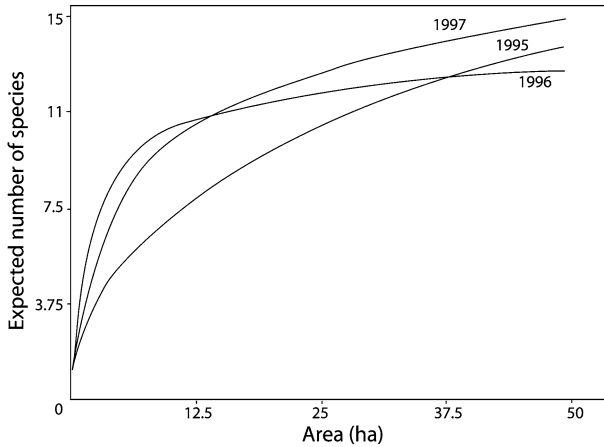
Fig. 2 Rarefaction analysis estimating the expected number of grassland resident bird species per sampled area in tallgrass prairies at DNWR, Iowa, USA, 1995–1997. Different years plotted separately

(*Colinus virginianus*) ($F_2 = 3.9$, $P = 0.04$) and ring-necked pheasant (*Phasianus colchicus*) ($F_2 = 8.3$, $P < 0.01$), showed significant treatment–year interactions. Among these species, the density of bobwhite was 3.5 times higher ($t_8 = -3.2$, $P = 0.04$, Bonferroni corrected) and the density of pheasant 10 times higher ($t_8 = -3.0$, $P = 0.05$, Bonferroni corrected) on burned sites two years after burning compared to untreated sites. Densities were not different between treatments in earlier years, suggesting a time lag in the effect of burning in these species.

Discussion

Factors affecting responses in plant communities

The low diversity and abundance of plant species in tallgrass prairie fragments at DNWR are, in part, reflections of site-specific management histories. Initial re-establishment of grasslands at DNWR did not use the historical array of plant species that were typical of tallgrass prairie systems before settlement and cultivation, but only a small number of native species that could be inexpensively and efficiently planted. Prairie fragments at DNWR retain the influence of their initial plantings, and their low levels of species diversity also are representative of the depauperate condition of their surrounding landscape, one that lacks larger prairie remnants and diverse seed banks.

In our study, only *Cassia fasciculata*, a N-fixing legume, was exclusive to burned sites. Burning is known to increase forb biodiversity in N-limited systems by benefiting species of N-limited legumes through increased N availability in soils (Dudley and Lajtha 1993; Towne and Knapp 1996). The fact that burning added only one such species in our study is yet further evidence of the impoverished state of local seed banks.

Factors affecting responses in bird communities

Like the simplified communities of plants they inhabited, communities of native grassland-dependent birds on burned and unburned sites also possessed low species diversity. Patterns of spatial distribution and community organization on these sites supported the long-held hypothesis that the persistence and success of species in grassland habitat are related directly to the ecological differences (i.e., niche separation) between them (Cody 1968). The small size of these fragments, coupled with their low diversity of plant species, probably reduced habitat heterogeneity and contributed to reduced opportunity for niche separation, resulting in low species diversity.

At DNWR, area sensitivity, another important constraint on bird species diversity, might be mediated by density. Rarefaction analysis suggested that, at high densities (1996), most species were present on larger sites. This pattern of occupancy is consistent with predictions of the Ideal Free Distribution theorem (Rosenzweig 1991), which asserts that strength of habitat selection declines with increasing population density. Our results also are consistent with species-specific null models which suggest that, at low densities, a species' probability of occurrence decreases with decreasing field size (Horn et al. 2000). Such considerations might explain why rarefaction analysis showed fewer species associated with areas in the size range of our sites (3–10 ha) in a year of low population densities (1995) than in years of higher population densities (1996 and 1997).

Both species of upland game birds, northern bobwhite and ring-necked pheasant, experienced increases in density on burned sites by the second year after burning, but not before. These species are ground-nesters that begin incubating in April at DNWR, before spring burning occurred. Thus, spring burning in 1995 destroyed some nests, making its immediate effects detrimental. However, increased densities on burned sites after 2 years suggested that longer-term effects of burning might be beneficial. Although we cannot identify the mechanism of such increases given the apparent similarity of plant communities on burned and unburned sites, such patterns of population change should alert managers to the fact that the effects of burning on individual species might involve time lags that would not be detected without ongoing, longer-term monitoring.

Management implications

Plant communities

Our results demonstrated that, in plant communities, many small sites do make cumulative contributions to landscape biodiversity. Thus, there is value in managing such small sites effectively and intensively, and they should not be neglected in an overall strategy of plant conservation.

At DNWR, most grass cover on burned sites was contributed by just five species of C_4 grasses (*Andropogon gerardii*, *Bouteloua curtipendula*, *Panicum virgatum*, *Schizachyrium scoparium* and *Sorghastrum nutans*). These species benefit from early spring burning on these sites, but their resulting dominance does not encourage the establishment of more diverse and historically representative plant communities. If historic plant diversity is the normative management goal, managers might achieve

greater success through summer burns rather than fall or spring burns (Howe 1994a, b), and by using burns less frequently rather than more frequently.

Bird communities

Given that burning had no effect on species richness and diversity of resident grassland birds, species–area relationships might be more important determinants of these variables, especially given the demonstrated sensitivity to area in some species (Herkert 1994a, b; Vickery et al. 1994; Swengel 1996). It is likely that grassland sites need to be more than 50 ha in size, preferably at least 200 ha, if they are to support a high level of avian biodiversity (Vickery et al. 1994). Field sparrow, the only species suggesting sensitivity to treatment in our study, also was the only species known to be insensitive to area (Vickery et al. 1994). Managers might have greater success in enhancing bird species diversity, as well as increasing densities of individual species, by acquiring and restoring larger sites rather than investing in intensive management of smaller sites.

Although our results demonstrated that many small sites can contribute to increased landscape diversity of plant communities, the collective contribution of small sites was less pronounced for bird communities, particularly at higher population densities. At DNWR, it appears that adding additional small sites would provide little increase in total bird species diversity. If this is the case, managers should use small fragments to benefit selected species while simultaneously working to acquire and restore sites > 200 ha in size in order to develop larger and more diverse avian communities.

References

Askins RA (1993) Population trends in grassland, shrubland, and forest birds in eastern North America. Curr Ornithol 11:1–34
Cody ML (1968) On the methods of resource division in grassland bird communities. Am Nat 102:107–147
Collins SL (2000) Disturbance frequency and community stability in native tallgrass prairie. Am Nat 155:311–325
Collins SL, Knapp AK, Briggs JM, Steinauer EM (1998) Modulation of diversity by grazing and mowing in native tallgrass prairies. Science 280:745–747
Cully JF Jr, Michaels HL (2000) Henslow's sparrow habitat associations on Kansas tallgrass prairie. Wilson Bull 112:115–123
Dudley JL, Lajtha K (1993) The effect of prescribed burning on nutrient availability and primary production in sandplain grasslands. Am Midl Nat 130:286–298
Dyer DP (1978) An analysis of species dissimilarity using multiple environmental variables. Ecology 59:117–125
Edgington ES (1995) Randomization tests, 3rd edn. Marcel Dekker, New York, USA

Herkert JR (1994a) The effects of habitat fragmentation on Midwestern grassland bird communities. Ecol Appl 4:461–471

Herkert JR (1994b) Status and habitat selection of the Henslow's sparrow in Illinois. Wilson Bull 106:35–45

Horn DJ, Fletcher RJ Jr., Koford RL (2000) Detecting area sensitivity: a comment on previous studies. Am Midl Nat 144:28–35

Howe HF (1994a) Response of early- and late-flowering plants to fire season in experimental prairies. Ecol Appl 4:121–133

Howe HF (1994b) Managing species diversity in tallgrass prairie: assumptions and implications. Conserv Biol 8:691–704

International Bird Census Committee (1970) Recommendations for an international standard for a mapping method in bird census work. Swedish National Scientific Research Council, Stockholm. Bull Ecol Res Comm 9:49–52

Janzen DH (1983) No park is an island: increase in interference from outside as park size decreases. Oikos 41:402–410

Krebs CJ (1989) Ecological methodology. Harper Collins, New York, USA

Leach MK, Givnish TJ (1996) Ecological determinants of species loss in remnant prairies. Science 273:1555–1558

Noss R, Csuti B (1992) Habitat fragmentation. In: Meffe GK, Carroll CR (eds) Principles of conservation biology. Sinauer Associates, Sunderland, Massachusetts, USA, pp 269–304

Pielou EC (1975) Ecological diversity. Wiley, New York, USA

Rosenzweig ML (1991) Habitat selection and population interactions: the search for mechanism. Am Nat 137:S5–S28

Samson F, Knopf F (1994) Prairie conservation in North America. BioScience 44:418–421

Shannon CE, Weaver W (1949) The mathematical theory of communication. University of Illinois Press, Urbana, USA

Swengel SR (1996) Management responses of three species of declining sparrows in tallgrass prairie. Bird Conserv Int 6:241–253

Towne EG, Knapp AK (1996) Biomass and density responses in tallgrass prairie legumes to annual fire and topographic position. Am J Bot 83:175–179

Vickery PD, Hunter ML, Melvin SM (1994) Effects of habitat area on the distribution of grassland birds in Maine. Conserv Biol 8:1087–1097

Yahner RH (1988) Changes in wildlife communities near edges. Conserv Biol 2:333–339

Population trends and spatial synchrony in peripheral populations of the endangered Lesser grey shrike in response to environmental change

David Giralt · Francisco Valera

Abstract Regional synchronization in species dynamics as well as particular ecological and demographic characteristics of peripheral populations poses special challenges for conservation purposes, particularly under the current scenario of global climate change. Here, we study the population trend and spatial synchrony of several peripheral populations of the endangered Lesser grey shrike *Lanius minor* at the western limit of its breeding range (southern France and northeast Spain). In an attempt to ascertain the effect of environmental change on the decline of the species we also look for evidence of climate changes in the breeding and wintering area of this shrike and related effects on vegetation by using the normalized difference vegetation index (NDVI). We found that the interannual fluctuations of the peripheral populations in France and Spain are strongly correlated, therefore suggesting that their decline can be under the influence of a common factor. We obtained clear evidence of climatic change (an increased thermal oscillation) in one peripheral population that could have resulted in a decrease of the NDVI index in the area. Our study finds correlational evidence that climatic variables in the breeding area may account for fluctuations in abundances of some populations and that environmental conditions experimented by some population could influence the fate of the neighboring populations. Our results indicate that the studied peripheral populations are spatially synchronized, so that conservation efforts should be applied at a large-scale encompassing all the isolated populations at the western border of the range of the species in the Mediterranean area.

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Introduction

Understanding the spatial distribution of organisms is a crucial issue in population
dynamics (Tilman and Kareiva 1997) and has important consequences for more
applied sciences such as conservation biology, where strategies for long-term species
conservation depend on present and future patterns of abundance (Lawton et al.
1994; Brown et al. 1995; Mehlman 1997; Williams et al. 2003). One major topic in
this field is the comprehension of the abundance distribution within the range of a
given species and the roles of density-dependent and density-independent processes
in determining the variability of population abundances (Hengeveld and Haeck
1982; Brown 1984; Royama 1992; Mehlman 1997; Williams et al. 2003). It is well-
known that peripheral populations, frequently relatively small and isolated from
central populations (Lawton 1993; Lesica and Allendorf 1995), are likely to expe-
rience different regimes of natural selection than central ones, since the relative
importance of abiotic and biotic factors on distribution patterns and population
limitation are likely to change according to the position within the geographical
range (Randall 1982; Hoffmann and Blows 1994; Brown et al. 1995; Williams et al.
2003). There is evidence supporting the fact that environmental changes and abiotic,
density-independent factors, like weather, have a higher influence on demographic
rates and produce greater fluctuations in peripheral populations than in central ones
(Hoffmann and Blows 1994; Brown et al. 1995; Curnutt et al. 1996; Williams et al.
2003), probably because closeness to the edge of range usually indicates poorer
environmental conditions for a species (Brown 1984; Brown et al. 1995). Thus, in the
present scenario of climate change (Parmesan et al. 1999; Hughes 2000), we would
expect peripheral populations of organisms to be under a stronger influence of
environmental changes than populations closer to the core of their range.

   An additional factor of uttermost importance in the conservation of peripheral
populations and/or of rare or endangered species is spatial synchrony in the
dynamics of local populations (Kendall et al. 2000). Spatial synchrony refers to
coincident changes in the abundance or other time-varying characteristics of geo-
graphically disjunct populations (Liebhold et al. 2004). Evidence for widespread
spatial synchrony in population fluctuations has been found in a variety of organisms
(Paradis et al. 2000; Kendall et al. 2000), and some studies have found that weather
is a likely candidate as a synchronizing factor (Paradis et al. 2000; Williams et al.
2003).

   Climate per se is a major determinant of geographical distribution for many
organisms and recent climate warming has been shown to affect the distribution of
different species (Thomas and Lennon 1999; Parmesan et al. 1999; Hughes 2000).
However, the potential impact of a change in environmental suitability on abun-
dance within the range of a given species has received relatively little attention
(Mehlman 1997; Williams et al. 2003). This is partly because there is generally little
comprehensive information on the distribution of abundance within the range of a
species and because long-term estimates of densities over wide geographic areas are
uncommon (Williams et al. 2003).

In this paper, we study the pattern of population synchrony of three peripheral populations of the Lesser grey shrike *Lanius minor*, and the relationship between population fluctuations and weather. The Lesser grey shrike is highly endangered throughout Europe, having declined markedly in abundance and range in the last decades (Lefranc and Worfolk 1997). Relict populations of the species in the western limit of its breeding range are known to exist since long (Lefranc 1995). Such peripheral populations, now restricted to very small areas in southern France and northeast Spain, have been reported to decline in the last decades (Lefranc 1995; Giralt and Bota 2003, Giralt 2004). Overall, there is almost no information on which factors may be responsible for the general decline of the species (Lefranc and Worfolk 1997; Krištín et al. 2000). Some authors have pointed out loss of habitat (agricultural intensification) and adverse weather (Lefranc 1995, 1997; Lefranc and Worfolk 1997; Isenmann and Debout 2000) whereas others suggest that adverse circumstances on the wintering grounds or during migration may account for the decline (Herremans 1997a, 1998a). However, no specific work has investigated the causes underlying the decrease of the species. Whereas there is not much information on this shrike (but see Lefranc 1995; Herremans 1997a; Lefranc and Worfolk 1997; Isenmann et al. 2000; Krištín et al. 2000), the distribution of abundance of the species in its westernmost range and changes in the last decade are available, what provides a valuable opportunity for studying population fluctuations and general patterns of changes in abundance in response to environmental changes. Therefore we aim to: (i) evaluate the spatial scale of synchrony in fluctuations of several peripheral populations of this species, (ii) assess the role of climate on the decline of this shrike in an attempt to contribute to the conservation of this particular species. Additionally we aim to illustrate the challenges associated to the conservation of peripheral populations of endangered birds.

Methods

Study species and study area

The Lesser grey shrike is a socially monogamous long-distance migratory passerine whose breeding range is limited to warmer parts of Eurasia, spreading over 6,000 km from west to east (Cramp and Perrins 1993). The westernmost points reached by this species lie in southern France and northeast Spain (Lefranc and Worfolk 1997), at the farthest extreme of its migratory route. This shrike produces a single brood per season, although replacement clutches can be produced after nest failure. Birds arrive on the breeding grounds in Europe during May. In the study area the main egg-laying activity takes place from late May to early June and the main fledging period encompasses from late June to early July (Isenmann and Debout 2000; pers. obs.).

Two main breeding areas existed in Spain until recently (Giralt and Bota 2003). The breeding area in Girona (42° 16′ 42.84″ N 3°07′ 21.66″ E, Catalonia) lay in the Natural Park of Aiguamolls de l'Empordá (protected area since 1983) and its periphery, where the species bred for the last time in 2001 (Fig. 1). The breeding area in Lleida (Catalonia) is 220 km southwest from the first one, and consists of two

nuclei 5,5 km apart from each other (Fig. 1). A third, relict nucleus remains in
Aragón (35 km west of Lleida).

The breeding population of the study species in France is located in two main
breeding areas, Montpellier (43°30′ 44.66″ N 3°39′ 15.26″ E, Hérault) (Isenmann
et al. 2000) and Aude (43°15′ 53.48″ N 3°08′ 51.78″ E, departments of Aude and
Hérault, Bara 1995), 60 km from each other and 180 and 120 km from the breeding
nucleus in Girona, respectively (Fig. 1). Some isolated pairs (3 in 2002 and 1–2 in
2004) still breed in Vaunage (Gard) (Labouyrie 2003; pers. comm.).

The non-breeding range of the Lesser grey shrike is about one tenth of the size of
its breeding range. During the non-breeding season, the world population concen-
trates in the southern African thornbelt, mainly in the Kalahari basin (Herremans
1997a, 1998a), spreading mostly over Botswana, Namibia and north of South Africa.
Birds occupy their final non-breeding destination in January–March and almost all
individuals have disappeared from the winter quarters by the end of April (Herre-
mans 1997b).

Abundance data

Data on the past distribution of the species in the Iberian Peninsula have been
collected from old and recent literature and personal communications. Long-term
data on breeding population come from Girona where the population has been
monitored during 1989–1997 (except 1992) by the staff of the Natural Park, and by
D.G. from 1998 to 2004. Data on the current distribution of the species in Lleida
have been collected on the basis of censuses performed by D.G. during the entire
breeding season (15 May–10 July) for the period 2001–2004.
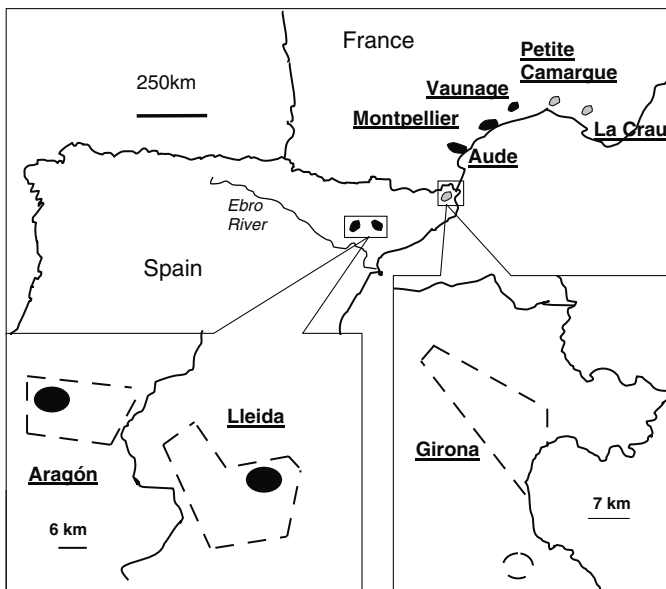

Fig. 1 Contraction of the breeding range of the French and Spanish populations of Lesser grey
shrike. Current breeding areas (in black), past breeding locations (shaded areas) and contraction of
the range (discontinuous lines) in the study areas are shown

                                        [18]

Information about the two main areas in France was collected from Bara (1995), Isenmann et al. (2000), Donck and Bara (2001) and Rufray and Rousseau (2004) and kindly completed by Isenmann, Lefranc and Rufray. Censuses in the population in Aude started in 1992 (Bara 1995). The breeding population in Montpellier was discovered in 1995 (Béchet et al. 1995) and censused from that year onwards.

Abundance data from Spain and France are the result of censuses specifically undertaken to monitor the breeding population of this species. They were performed from territory establishment until fledging. Thus, such information reflects reliable actual counts.

Population trends, fluctuations and spatial synchrony

We studied the occurrence and intensity of population synchrony by correlating the fluctuations in the time series of the French and the Spanish (only Girona) populations (Paradis et al. 2000). We used the software program TRIM (Pannekoek and van Strien 2003), designed to analyze time series of counts with missing observations by using Poisson regressions, that produces estimates of yearly indices and trends.

First, we studied the trend of the French population (Aude and Montpellier) from 1993 to 2004 (two missing data—1993 and 1994—for the population in Montpellier) and of the population in Girona from 1993 to 2002 (although the species bred there for the last time in 2001 we assigned 1 pair to 2002 to allow a better comparison between both populations—see below). For these purposes we used a linear trend model with stepwise selection of change-points. Change-points are moments in time (i.e. years) where the slope parameter changes. This model provides an overall trend as well as selects specific time points (i.e. years) when the slope parameters differ significantly from the ones obtained before and after that time point. Possible violations to the assumption of Poisson distributions due to overdispersion or serial correlation were corrected with the methods implemented in TRIM.

Then, we explored whether the fluctuations of the Spanish (Girona) and French population (Aude and Montpellier) are interrelated by examining the relationship between the annual departures from the long-term trends of each population. For this we first calculated the trend for each population from 1993 to 2002 and the yearly deviations from each linear trend by using the time effects model implemented in TRIM. Since null values are not admitted when using the time effects model we restricted the study period until 2002 (breeding in Girona did not occur from 2002 onwards, see results), and considered one pair to breed in 2002 after checking that the linear trend obtained was similar to the one estimated with the linear trend model run with zero breeding pairs in 2002. In contrast to the linear trend model this one calculates separate parameters for each year and estimates yearly deviations from the linear trend. This model provides the best estimates for deviations of the general trend (van Strien, pers. comm.). Finally, we correlated the yearly deviations from the linear trends found in France and Girona.

Meteorological variables

We used meteorological data from the meteorological stations closest to each breeding population: Mauguio–Montpellier (Montpellier), Béziers (Aude) and Aiguamolls de l'Empordà (Girona), all of them less than 15 km from the respective breeding sites. Weather records consisted in daily rainfall (mm) during

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January–July for the period 1989–2001 for Girona and maximum and minimum daily temperatures (°C) during May–June (encompassing most of the breeding period of the study species) for the period 1989–2002 for Aude, Montpellier and Girona. Daily thermal oscillation was calculated as the difference between the latter variables. Mean temperatures were calculated as the average of daily maximum and minimum temperatures.

Rainfall in Botswana and Namibia (October–February) was also gathered from the Tyndall Center for Climate Change Research (data set TYN CY 1.1, Mitchell et al. 2002). Since rainfall varies widely in and around the Kalahari we compared seasonal rainfall (October–February) during the years 1989–2000 with the mean rainfall for those months in the same areas for the larger period 1902–2000 (data set TYN CY 1.1). We got an average of 502.93 mm. We defined a threshold value of 150 mm below and above the long-term mean to classify years of the period 1989–2000 into 'low', 'average' and 'high' rainfall years (thresholds of 352.93 and 652.93 mm, respectively). Similar classifications have been applied in other studies (see, for instance, Wiegand et al. 1999, Tews and Jeltsch 2004).

Normalized difference vegetation index (NDVI)

The amount and vigor of vegetation at the land surface was estimated by means of the NDVI. This index, based on satellite images indicating the condition of rainfall-dependent vegetation in time, is strongly correlated with the fraction of photosynthetically active radiation absorbed by vegetation (see Asrar et al. 1984; Prince and Justine 1991; Myneni et al. 1997 for more details about the index and Sanz et al. 2003 for a similar use of the index).

NDVI data corrected by surface topography, land-cover type, presence of clouds and solar zenith angle were provided by Clark Labs in IDRISI format as world monthly images at spatial resolution of 0.1° in a 0–255 scale values. Using IDRISI32 software, we obtained mean NDVI values for the period May–June (at 1-month interval) from 1988 to 2000 for the square areas sized 0.25° occupied by breeding populations in Montpellier (E 3°30′–3°45′ N 43°25′–43°40′), Aude (E 3°–3°15′ N 43°10′–43°25′) and Girona (E 3°–3°15′ N 42°10′–42°25′) and for the period January–March (when most birds occupy their final wintering destination) from 1988 to 2000 for the wintering area (E 18°–28°, S 20°–27°). The selected wintering area matches the Kalahari basin defined by Herremans (1997a, 1998a) as the core area for the Lesser grey shrike.

Statistical analyses

Separate stepwise multiple regressions were performed to determine the effect of climatic variables (thermal oscillation) and NDVI indexes on the population size of each of the three peripheral areas. Thermal oscillations in the three study areas during May–June were the independent variables for the first set of analyses. For the second set we used the mean NDVI index during May–June of each study area as well as the mean NDVI index for January-March in the wintering grounds in Africa. Parametric tests were used where the assumptions for normality were met. In some cases transformations were used to meet the requirements for normality. Otherwise non-parametric tests were used. Statistical analyses were carried out with the

STATISTICA 6.0 package (StatSoft Inc. 2001). Unless otherwise stated means and standard errors are offered and two-tailed tests used.

Results

Contraction of the breeding range

The geographic range of the species at the southwestern limit of its distribution has contracted dramatically. The breeding nucleus in Girona, that held in 1989 around 50% of the estimated breeding population in Spain, contracted progressively until it became extinct in 2002 (Fig. 1). Although the breeding population in Lleida seems relatively stable during the last years (1.4, 2.4, 1.4 and 2.3 breeding pairs/km^2 for 2001–2004) the breeding range has contracted ca. 50% in last decade due to the progressive disappearance of breeding sites to the south and west of the current breeding area (Fig. 1). As a whole, the contraction of the range in Catalonia (Lleida and Girona) has been estimated at –68% between 1984 and 2002 (Giralt 2004).

The breeding area in Aragón, with ca. 2–7 pairs in the last 3 years, has also contracted during the last decade after disappearance of the easternmost breeding sites (Giralt and Bota 2003; Albero and Rivas, pers. comm.) (Fig. 1).

The French population has also declined in range due to the loss of breeding localities during the 90s in Gard (Petite Camargue) and Bouches du Rhône (La Crau), a stronghold of the species in the 70s (Lefranc 1999; Donck and Bara 2001; Labouyrie 2003) (Fig. 1). More recently, a 50% reduction of the number of pairs in the main nuclei (Aude and Montpellier) since 2002 (Rufray and Rousseau 2004) has contributed to the contraction of the range (Fig. 1).

Population trend and spatial synchrony in peripheral populations

Breeding populations of the Lesser grey shrike have been decimated along the southwestern range of its distribution (Fig. 2).

The decline of the species in France (Montpellier and Aude) fits a linear model (Likelihood ratio = 1.83, df = 10, $P = 0.99$) with a significant decreasing slope of 8.25% per year for the period 1993–2004 (Overall Multiplicative Slope imputed with intercept = 0.917, SE = 0.0069, $P < 0.05$). Particularly marked decreases in the trend occur between 1997–1998, 1999–2000, and 2001–2002 (Tables 1 and 3). During 2002–2004 there was a constant yearly decrease of 75% that has resulted in the lowest ever numbers of the French population (Fig. 2).

The decline of the Spanish population in Girona during the period 1993–2002 also fits a linear model (Likelihood ratio = 0.03, df = 1, $P = 0.85$) with a significant decreasing slope of around 14% per year for the period 1993–2002 (Overall Multiplicative Slope imputed with intercept = 0.856, SE = 0.013, $P < 0.05$) (Fig. 2). Similar to the French population, this tendency is not constant and several significant change-points can be distinguished (Table 2). Sharp decreases occur between 1993–1994, 1995–1996, and 1997–1998 whereas increases occurred between 1994–1995, 1996–1997 and 1998–1999 (Tables 2 and 3). During 1999–2001 there was a constant yearly decrease of 76% that, in fact, led to the extinction of the species in Girona from 2002 onwards (Table 3, Fig. 2).