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A Comprehensive Approach to Identifying Monitoring Priorities of Small Landbirds on Military Installations

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ABSTRACT / Military installations provide important native habitat for songbirds, including many species that have experienced population declines in recent decades. As part of the Land Condition Trend Analysis (LCTA) program to monitor animal populations on military lands, we surveyed small (<250 g) breeding landbirds on 60 permanent plots on the Fort Riley Military Installation in northeastern Kansas from 1991 to 2002. During this period, species richness averaged 39.0 species (SE 0.9)/year and mean species richness per plot ranged from 3.6 species (SE = 0.2)/plot (1999) to 7.5 species (SE = 0.3)/plot (1992). Turnover (the appearance and disappearance of species on all plots from one year to the next) ranged from 5 species (2000–2001) to 16 species (1992–1993) and was driven primarily by turnover of woodland species. We developed an index of relative difference (C) to evaluate relative trends of local populations and found that 25 species declined, 15 species increased, and 7 did not change. Based on migration assemblages, more resident species (6 of 10) and more short-distant migrants (9 of 12) decreased than long-distance migrants (10 vs. 11). Our analysis of major vegetation communities on plots showed few changes in the quantity of habitats (grassland vs. woodlands) during the study. Our results indicate that Fort Riley provides important habitats for many landbirds, particularly those that require grasslands for breeding. Several species exhibited local declines when compared to the regional Breeding Bird Survey routes. We offer an approach that evaluates population changes of small landbirds and provides objective inputs for conservation directives. These can be adopted easily for use on military installations (that use LCTA), parks, and wildlife refuges that have data from annual breeding bird surveys.

Many songbirds that breed in North America have undergone population declines in recent years, particularly those that nest in shrublands and grasslands (Askins 1993; Vickery and others 1999; Vickery and Herkert 2001). Populations are influenced by many factors, yet habitat loss and degradation (e.g., fragmentation, succession to unsuitable habitat) are believed to play important roles in structuring populations (Sherry and Holmes 1995; Böhning-Gaese and others 1993; Askins 2001). On the breeding grounds, habitat loss and degradation can reduce reproductive success via reduced food supplies (Burke and Nol 1998; Zanette and others 2000), elevated rates of nest predation (Donovan and others 1995; Hoover and others 1995), and increased brood parasitism by the brown-headed cowbird (*Molothrus ater*; Robinson and others 1995a, 1995b). Depressed reproductive success on the local level can, in turn, influence population recruitment at the landscape scale. Thus, proper conservation and management of breeding

KEY WORDS: Index of relative difference; Landbirds; LCTA; Military installations; Population monitoring; Species richness

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habitat is critically important to maintain viable populations (Martin 1992).

The Department of Defense is the second largest land steward in the United States and oversees 10.4 million ha of land, much of which is managed as wildlife habitat (Cohn 1996). Military installations have opportunities to conserve songbird habitats because they typically contain large, contiguous areas that comprise a diversity of habitats. Yet, the contributions of military installations to the conservation of songbirds are poorly documented. Little has been done to understand how military lands contribute to regional populations and how activities on military reservations influence local populations, even though threatened and endangered species have been studied on military installations (e.g., Barber and Martin 1997; Sockman 1998; Bowman and others 1999). The US Army Land Condition Trend Analysis (LCTA) program was developed in the late 1980s and implemented variously on most Army installations in 1991, with the intent of inventorying and monitoring natural resources (Diersing and others 1992). Some alternatives to the sampling methodology of the LCTA bird protocol have been recommended, but the basic guidelines appear to be adequate for monitoring the common species (Cully and Winter 2000). LCTA surveys, therefore, could be providing critical baseline data on songbird populations on these large, contiguous areas.

We used a 12-year LCTA dataset from the Fort Riley Military Installation (hereafter Fort Riley) in northeast Kansas to evaluate population trends of breeding landbirds. We begin by describing temporal changes in the avian community and evaluating populations for the dominant species that breed in Fort Riley. We then consider if vegetation succession could have influenced recent population changes. Because local dynamics is ultimately influenced by regional processes, we used the North American Breeding Bird Survey (BBS) data to relate BBS population trends to trends of species breeding in Fort Riley. We outline the computing techniques we used to evaluate small landbird populations so that these techniques can be adopted by managers of military lands, parks, and wildlife refuges.

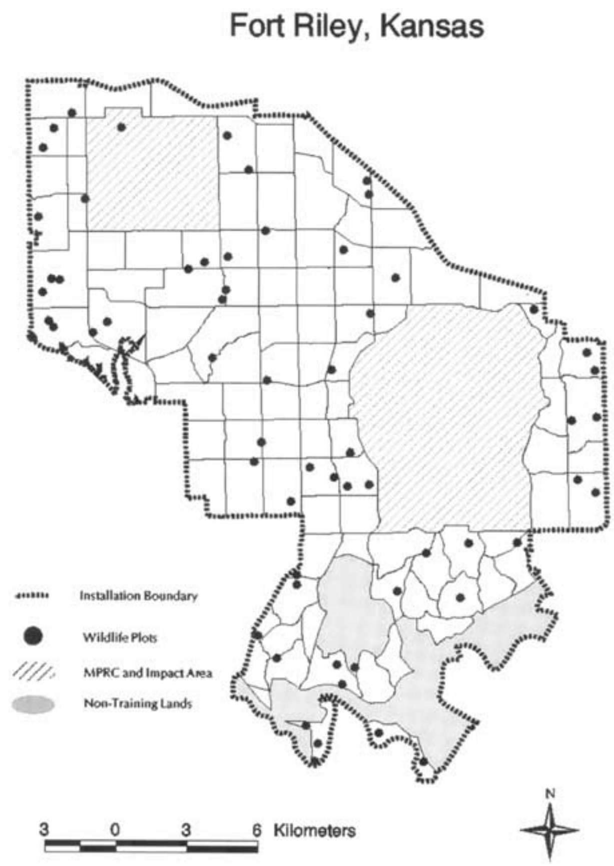


Figure 1. Distribution of bird survey plots ($n = 60$) at Fort Riley, Kansas, relative to nontraining and restricted use areas (MPRC and Impact Area) off limits to nonmilitary personnel for security and safety reasons.

Methods

Fort Riley is a 40,273-ha training area of the US Army located in Clay, Geary, and Riley counties in the Flint Hills of northeastern Kansas (39°15′N, 96°50′W; Figure 1). The Flint Hills region encompasses over 1.6 million ha covering much of eastern Kansas from near the Kansas–Nebraska border south into northeastern Oklahoma and contains the largest remaining areas of unplowed tallgrass prairie in North America (Knapp and Seastedt 1998). The climate is characterized by hot summers and cold, dry winters. Mean monthly temperatures range from −2.7°C in January to 26.6°C in July. Annual precipitation averages 83.5 cm, with 75% of precipitation occurring during the growing season (Hayden 1998).

Avian habitats on Fort Riley are diverse. There are three major vegetation communities: grasslands (~32,200 ha), shrublands (~6000 ha), and woodlands (~1600 ha). Grasslands are dominated by Indiangrass (*Sorghastrum nutans*), big bluestem (*Andropogon gerardii*), switchgrass (*Panicum virgatum*), composite dropseed (*Sporobolus compositus*), and little bluestem (*Schizachyrium scoparium*), with other annual grasses and forbs occurring at lower abundances. Shrublands are dominated by coralberry (*Symphoricarpos orbiculatas*), smooth sumac (*Rhus glabra*), leadplant (*Amorpha *canescans*), white sagebrush (*Artemisia ludoviciana*), rough-leaved dogwood (*Cornus drummondii*), and a mixture of grasses and forbs. Shrublands occur along woodland edges and in isolated patches in grassland areas. Woodlands typically occur along riparian lowlands and are characterized by chinquapin oak (*Quercus muhlenbergii*), bur oak (*Q. macrocarpa*), American elm (*Ulmus americana*), hackberry (*Celtis occidentalis*), and black walnut (*Juglans nigra*).

Military training in Fort Riley includes field maneuvers, combat vehicle operations, mortar and artillery fire, small-arms fire, and aircraft flights. Wildfires resulting from training activities can occur during any season on the installation, but typically occur during spring. Management activities include mowing, chemical control of noxious weeds, prescribed burning, and small-scale (<50 ha/year) timber harvest.

We established 60 permanent plots across Fort Riley (Figure 1) using a stratified sampling design where plots were stratified by land cover and soil types. The number of plots selected was proportional to the strata areas based on 1989 satellite images [see Diersing and others (1992) for details]; this resulted in 48 plots in grasslands and 12 in woodlands. All plots were placed >200 m from each other with the intention of producing uncorrelated observations (Ralph and others 1995). We surveyed each plot once per breeding season (i.e., late May–late June) using a limited-distance strip transect method (Verner 1985). During 1991–1998, a single observer walked a 100-m transect at a constant pace for 6 min and recorded all birds detected within a 1000-m radius of his current position (Figure 2a). During 1999–2002 surveys, a single observer walked along the same 100-m transect, centered on the plot, at a constant pace for 6 min and recorded all birds detected within an area bounded at 100 m perpendicular to the transect (Figure 2b). Although these techniques differ in their observational areas, they can be expected to yielded similar results because only presence–absence of species on each plot was considered in our analysis. Both techniques represent the ''line-out'' portion of the standard LCTA bird survey protocol (Tazik and others 1992). All surveys were conducted between sunrise and 1000 CDT when the wind was calm or low and there was no steady rain. The order in which plots were surveyed changed annually because of military training activities.

To examine population changes, we focused our analyses on small landbirds (<250 g; see Table 1 for scientific names of all species recorded ≥3 years). We examined community-level attributes of the avian community by calculating species richness and mean species richness/plot for each year (Magurran 1988). Species richness was calculated as the number of species observed over all plots in each year. Mean species richness/plot was calculated as the mean number of species/plot in each year. We computed turnover by calculating the appearance and disappearance of species on each plot from one year to the next (Collins 2000). Appearance (A_j) is the number of species that were not observed in year j but were observed in year j + 1 and is computed as

$$A_j = \sum_{i=1}^{n} A_{ij}$$

where A_{ij} is the appearance of species on plot i and n_j is the number of plots surveyed common to years j and $j + 1$, $j = 1,2,\ldots, t - 1$, for t years of evaluations. Disappearance (D_j) was computed as the number of species that were observed in year j but not observed in year j + 1 and is computed as

$$D_j = \sum_{i=1}^{n} D_{ij}$$

where D_{ij} is the disappearance on plot i. Total turnover was calculated as $A_j + D_j$. Turnover/plot was calculated as

$$\frac{A_j + D_j}{n_j}$$

We assigned species to grassland, shrubland, or woodland nesting assemblages based on Zimmerman (1993). We classified the brown-headed cowbird as a grassland species because it was typically observed in grasslands. We assigned each species to one of three migration assemblages based on Ehrlich and others (1988). Residents were classified as those species present on Fort Riley year-round. Short-distance migrants were those species that winter mainly in the United States. Long-distance migrants were those species that winter south of the United States–Mexico border.

We excluded species observed <3 years from the individual species analysis because of difficulties that might arise assessing long-term population changes from a few years of data; this criterion eliminated 4 of 51 species observed. Chimney swifts (*Chaetura pelagica*) and swallows were excluded from population analyses because our observation methods were likely unsuitable for these aerial insectivores. Although we recorded the number of individuals observed on each plot during surveys, we based our analyses on presence–absence data for each species on each plot as a means of minimizing potential interobserver biases (five observers were used from 1991 to 2002) and because differ-

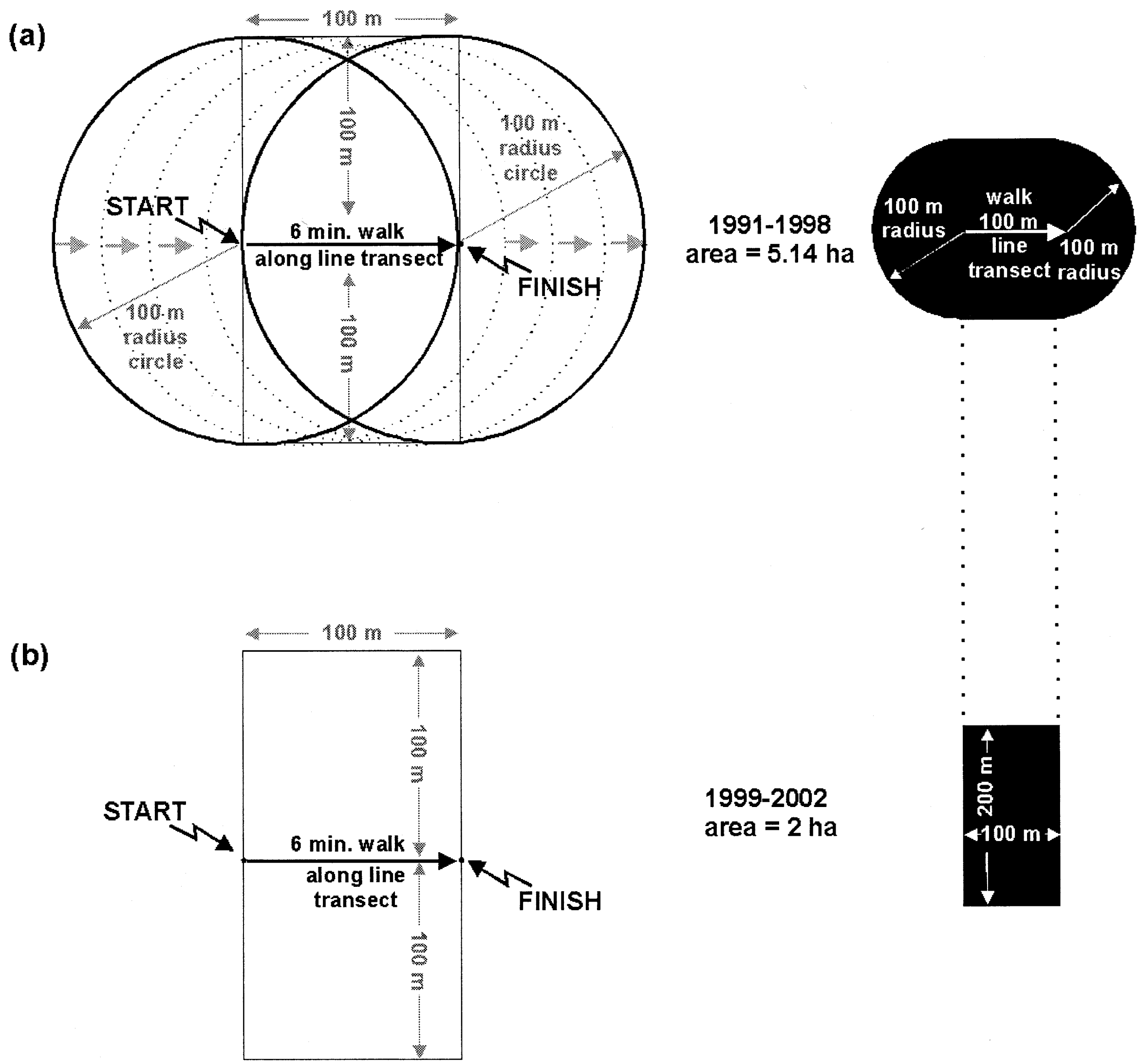


Figure 2. Schematics of bird survey protocol used from 1991 to 1998 (a) and 1999 to 2002 (b).

ences in the areas surveyed (Figure 2). Although presence–absence recordings might not be an optimal method to detect large, single-year changes in overall abundance, they do provide reasonably reliable indicators of long-term trends (Bart and Klosiewski 1989; Dunn and others 1996). Furthermore, we felt comfortable with our decision to use presence–absence for the analysis because the changes in plot sizes (larger: 1991–1998 vs. smaller: 1999–2002) yielded no instances where declines in richness were observed only between these two periods. Thus, we detected no bias in trends associated with a reduced plot size.

To assist in evaluating the status in species occurrences across years, we developed an index of relative differences:

$$C = \frac{\hat{p}_t - \hat{p}_1}{\sum_{j=1}^{t-1} |\hat{p}_{j+1} - \hat{p}_j|}$$

where

$$\hat{p}_j = \frac{1}{n_j} \sum_{i=1}^{nj} y_{ij}$$

and

$$y_{ij} = \begin{cases} 1, \text{ species observed at least once on plot } i \text{ in year } j \\ 0, \text{ otherwise} \end{cases}$$

The term $\hat{p}$ is the proportion of plots where at least one member of the species of interest was observed. The

Table 1. Birds observed at Fort Riley, Kansas, 1991–2002

Habitat assemblage (species)	Mean (SE)[a]	CV	Years[b]	*C* Fort Riley	*C* Regional[c]	PIF[d]
Grassland						
Dickcissel (*Spiza Americana*)	39.5 (1.6)	14.3	12	0.07	—[e]	26
Brown-headed cowbird (*Molothrus ater*)	35.1 (1.4)	14.2	12	−0.07	1.00	13
Grasshopper sparrow (*Ammodramus savannarum*)	31.2 (1.0)	11.4	12	0.23	0.12	22
Eastern meadowlark (*Sturnella magna*)	26.0 (3.2)	42.5	12	−0.35	−0.01	20
Eastern kingbird (*Tyrannus tyrannus*)	11.7 (1.9)	56.1	12	−0.24	−0.18	19
Red-winged blackbird (*Agelaius phoeniceus*)	7.8 (0.6)	27.1	12	−0.25	—[e]	15
Common yellowthroat (*Geothlypis trichas*)	7.3 (0.8)	39.1	11	0.28	0.02	15
Henslow's sparrow (*Ammodramus henslowii*)	6.1 (1.4)	76.9	11	0.02	0.31	28
Western kingbird (*Tyrannus verticalis*)	1.2 (0.3)	95.5	8	−0.06	0.01	20
Lark sparrow (*Chondestes grammacus*)	0.9 (0.3)	118.2	6	−0.20	0.43	17
Loggerhead shrike (*Lanius ludovicianus*)	0.5 (0.3)	200.0	3	−0.60	−0.42	19
Shrubland						
Northern cardinal (*Cardinalis cardinalis*)	9.8 (1.7)	58.5	12	−0.29	0.12	11
Brown thrasher (*Toxostoma rufum*)	8.5 (1.4)	55.5	12	−0.20	−0.18	22
Gray catbird (*Dumetella carolinensis*)	7.4 (0.7)	30.7	12	−0.03	0.11	17
Field sparrow (*Spizella pusilla*)	7.1 (1.1)	51.6	12	−0.31	0.06	23
American goldfinch (*Carduelis tristis*)	5.4 (0.7)	46.9	12	0.07	−0.17	15
Bell's vireo (*Vireo bellii*)	4.9 (0.7)	50.9	12	0.44	0.10	23
Woodland						
Blue jay (*Cyanocitta cristata*)	7.1 (2.1)	100.1	12	−0.28	−0.03	15
House wren (*Troglodytes aedon*)	6.7 (0.8)	39.1	12	0.00	−0.03	13
Yellow-billed cuckoo (*Coccyzus americanus*)	6.7 (1.6)	84.7	12	−0.22	0.10	21
Black-capped chickadee (*Poecile atricapillus*)	5.9 (1.1)	66.6	12	−0.26	−0.02	14
Common grackle (*Quiscalus quiscula*)	5.6 (2.0)	120.9	12	−0.15	0.00	13
Indigo bunting (*Passerina cyanea*)	5.3 (0.8)	53.8	11	0.00	−0.04	16
Tufted titmouse (*Baelophus bicolor*)	4.4 (0.8)	60.2	12	−0.04	0.29	14
Baltimore oriole (*Icterus galbula*)	4.1 (0.8)	67.2	12	−0.04	0.06	20
Red-bellied woodpecker (*Melanerpes carolinus*)	3.7 (1.5)	144.9	8	−0.37	0.13	16
Eastern towhee (*Pipilo erythrophthalmus*)	3.6 (0.6)	55.1	11	0.05	0.14	15
Great-crested flycatcher (*Myiarchus crinitus*)	2.8 (0.8)	103.0	8	0.08	0.09	21
Red-eyed vireo (*Vireo olivaceus*)	2.6 (0.4)	55.9	11	0.07	0.26	16
White-breasted nuthatch (*Sitta carolinensis*)	2.5 (0.5)	71.4	11	0.00	0.02	14
Yellow warbler (*Dendroica petechia*)	2.5 (0.4)	57.8	11	−0.05	−0.17	12
Orchard oriole (*Icterus spurius*)	2.1 (0.4)	72.2	10	0.12	0.22	22
Northern flicker (*Colaptes auratus*)	1.9 (1.0)	180.0	8	−0.65	−0.57	20
Carolina wren (*Thryothorus ludovicianus*)	1.8 (0.6)	120.5	8	0.02	0.14	13
Kentucky warbler (*Oporonis formosus*)	1.5 (0.3)	92.1	10	0.33	−0.18	22
European starling (*Sturnus vulgaris*)	1.5 (0.3)	77.8	9	−0.23	0.03	12
American robin (*Turdus migratorius*)	1.5 (0.4)	92.1	8	−0.20	−0.01	11
Blue-gray gnatcatcher (*Polioptila caerulea*)	1.4 (0.5)	114.4	8	0.18	0.06	17
Hairy woodpecker (*Picoides villosus*)	1.3 (0.3)	66.6	10	0.00	−0.10	15
Red-headed woodpecker (*Melanerpes erythrocephalus*)	1.3 (0.4)	102.3	7	−0.07	−0.29	25
Downy woodpecker (*Picoides pubescens*)	1.0 (0.3)	104.4	7	−0.20	−0.04	16
Northern parula (*Parula americana*)	0.8 (0.3)	123.6	6	−0.09	0.06	19
Eastern bluebird (*Sialia sialis*)	0.8 (0.3)	112.5	7	0.00	0.33	16
Black-and-white warbler (*Mniotilta varia*)	0.8 (0.2)	82.9	8	0.00	−0.13	18
Warbling vireo (*Vireo gilvus*)	0.7 (0.2)	97.7	7	0.11	0.00	18
Rose-breasted grosbeak (*Pheuticus ludovicianus*)	0.6 (0.3)	154.3	5	−0.40	−0.06	17
Willow flycatcher (*Empidonax traillii*)	0.3 (0.1)	180.9	3	−0.33	0.00	20

Note. The mean (SE) number of plots on which species were observed on each year during 1991–2002, coefficient of variation (CV), index of relative difference (*C*), and Partners in Flight (PIF) scores.

[a] n = 60 plots, except 1995 and 2000 (n = 59).

[b] Number of years where species was detected on ≥1 plot at Fort Riley. Only species observed ≥3 years are listed.

[c] All species listed were observed all survey years (n = 12) except American redstart (n = 4), black and white warbler (n = 7), and willow flycatcher (n = 11).

[d] Eastern tallgrass prairie region PIF scores were available for all species except the western kingbird for which the central mixed-grass prairie region PIF score was used.

[e] *C* value not generated because of division by zero.

numerator is the difference in the first and last proportions within the timeframe of interest. Any timeframe can be used. The denominator measures the variability of occurrences (through the proportions) within the timeframe and is not defined when all of the proportions are the same. The index has the range $[-1, 1]$, which gives bounds for interpreting index values. We use it only as a measure of consistency of proportional occurrences of a species over time and to assist in ordering species for possible increased monitoring or special management considerations.

The index is a measure of the difference of the proportions of the observed end-point occurrences over time. If the variability is relatively large as compared to the numerator (e.g., the yearly proportions noticeably fluctuate across years), the value of the index will be around zero; that is, there is enough inherent variability from year to year that a noticeable difference (if it exists) in the two end-point proportions of occurrences is not observed. Alternatively, if the variability is relatively small, then values of the index close to zero indicate little difference in the end-point proportions, or values of the index near -1 or near 1 indicate an observable difference in proportions at the two end points.

One can prove that the minimum and maximum values of the index are -1 and 1, respectively. If the value of the index is -1, then the proportions are monotonic and nonincreasing. For -1, all that is required is that the first end point be larger than the last end point, with at least two adjacent proportions with the same difference, and the remaining adjacent differences equal to zero. If the value of the index is 1, then the proportions are monotonic and nondecreasing. Values between -1 and 1 indicate the difference in the end-point proportions relative to the magnitude of the fluctuation in the sequence of observed proportions. In essence, the best use of the index is as a ‘‘flag’’ to indicate species that might warrant particular monitoring or management attention. For example, a species with an index value ‘‘near’’ -1 (an indication of decline in proportional occurrences) can be considered for more intensive monitoring and/or special management considerations. We resist recommending ‘‘cutoff’’ values or hypothesis tests, as the index is dependent on individual monitoring characteristics, such as sample sizes, probabilities of detection, and species’ behaviors.

We used loess (Cleveland 1985; James and others 1996) in conjunction with C to graphically assess changes in population trends. A smoothing parameter of $f = 0.5$ was selected because preliminary trials with this dataset indicated that this value provided adequate smoothing without distorting the underlying pattern in the data among species whose population trajectories differed markedly (unpublished data). The selection of f can be made using, for example, cross-validation, but these were not available in the software we used. Although we present a subset of the loess plots, plots for all species can be accessed at http://www.ksu.edu/kscfwru/.

We used the North American Breeding Bird Survey (BBS) (Droege 1990) data to determine the regional species pool of potential colonists at Fort Riley and to calculate regional C estimates for each species observed ≥3 years at Fort Riley. BBS data from 1991 to 2002 were obtained from each BBS survey route ($n = 92$) within a 325-km radius of Fort Riley (encompassing routes in Kansas, Nebraska, Iowa, Missouri, and Oklahoma) (www.mp2-pwrc.usgs.gov/bbs/). We chose this radial distance because it represents, on a landscape scale, the diversity of habitats found in Fort Riley and, thus, the potential pool of colonizing individuals. Regional C estimates were calculated in the same manner as Fort Riley estimates using each survey route as a plot and species presence–absence records.

As one measure of relative conservation importance, we summed the seven parameter scores associated with breeding assigned by the Partners in Flight (PIF) prioritization plan for each species for which we analyzed population changes (Carter and others 2000). The rating for each species can range from 7 to 35, with higher scores indicating higher conservation priority.

To determine if population trends were influenced by changes in vegetation composition, we compared color digital quadphotos [compiled to meet National Map Accuracy Standards (NMAS) for 1:12,000 products] of plots taken in 1991–1992 to color–infrared imagery (compiled to meet NMAS for 1:18,000 products at 0.5 m resolution) taken in 1998. On each plot, we used ArcView (Environmental Systems Research Institute, Inc.) to delineate a 100-m × 200-m grid that encompassed the core area sampled by both survey techniques. We divided each grid into 10-m × 10-m cells (200 cells in total), and classified each cell as woody or nonwoody vegetation based on the majority of vegetative cover it contained. In the few instances where cell classification via aerial images was equivocal, we used digital photos of the plots taken at ground level to aid vegetation classification. We then calculated the proportion of woody vegetation on each plot by summing the number of cells dominated by woody vegetation and dividing this sum by 200. We classified plots containing <30% woody cells as grassland plots, plots containing 30–70% woody cells as shrubland plots, and >70% woody cells as woodland plots. After we determined vegetation composition of all plots for both periods (i.e., 1991–1992 and 1998), we calculated

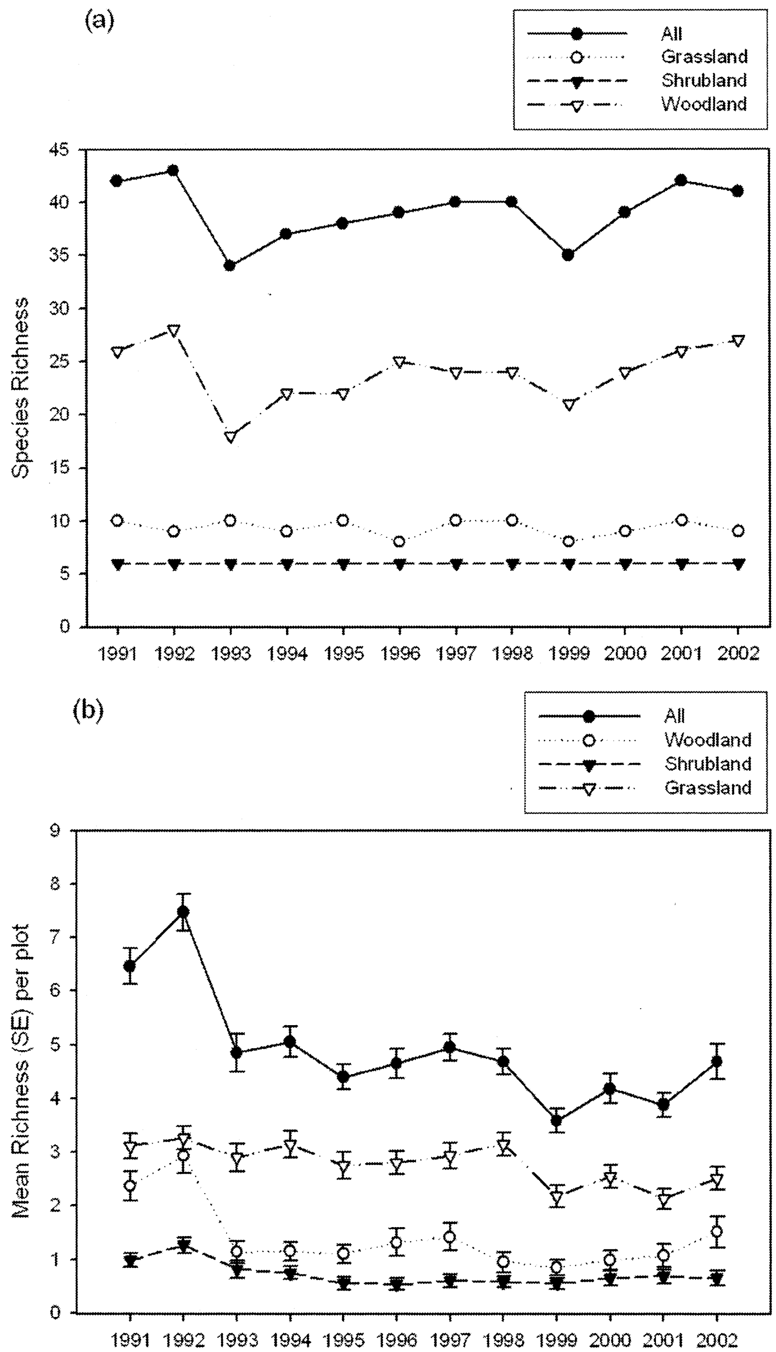


Figure 3. Small landbird species richness (a) and mean species richness/plot (b) at Fort Riley, 1991–2002, partitioned by breeding habitat association.

the difference between the periods as indices of vegetation changes on study sites. In 2000 using ground recognizance, each plot was inspected to determine if changes in classification had taken place since 1998.

Results

Estimates of species richness were similar among years (mean = 39.2, SE = 0.8), with the largest adjacent year decrease in richness occurring from 1992 to 1993 (Figure 3a). When partitioned by nesting habitats, woodland birds comprised the most species (18–28 species/year) followed by grassland birds (9–12 species/year) and shrubland birds (6 species/year, Table 1). Richness estimates for shrubland and grassland bird species varied less than richness of woodland species, which was largely responsible for changes in richness estimates among years (Figure 3a). Mean

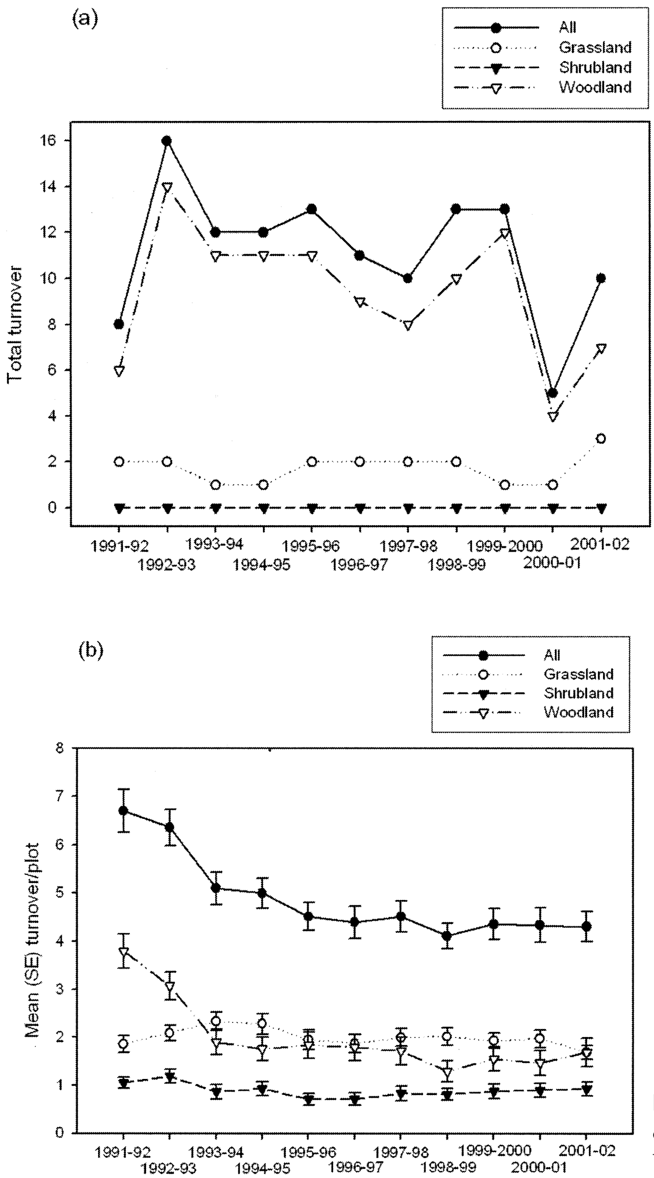
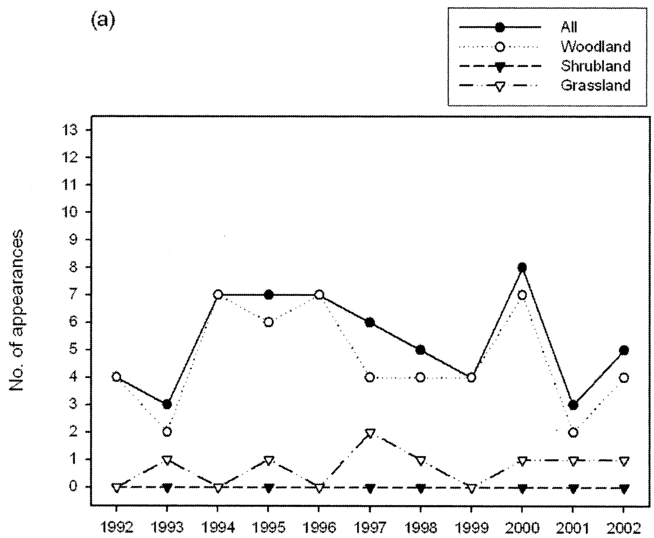


Figure 4. Total small landbird turnover (a) and mean turnover per plot (b) at Fort Riley, 1991–2002, partitioned by breeding habitat association.

species richness/plot ranged from 3.6 (SE = 0.2) in 1999 to 7.5 (SE = 0.3) in 1992. Annual changes were driven by fluctuations in the presence of woodland species (1995–2002) and, to a lesser extent, grassland species (1992–1992) (Figure 3b).

Total turnover ranged from 5 species (2000–2001) to 16 species (1992–1993) (Figure 4a). Total turnover was driven by appearances and disappearances of woodland species, which accounted for 70–92% of the total turnover during the 11 periods. In contrast, grassland species exhibited little total turnover (1–3 species/year), and shrubland species exhibited no turnover. Mean turnover/plot declined appreciably from 1991–1992 until 1993–1994, largely driven by turnover in woodland species (Figure 4b); from 1993–1994 through 2001–2002, the mean turnover/plot was consistently ≤ 2 for woodland and grassland species and ≤ 1 for shrubland species. Except for 1992–1993, when disappearances greatly exceeded appearances (12 vs. 3), the net change from year-to-year ranged from +3 to −4 species (Figure 5).

The proportion of plots on which a species was observed (hereafter, proportional occurrence) was largest for grassland species, as 5 species were observed on >12 plots/year, reflecting the dominance of grassland habitats (Table 1). The dickcissel was the most frequently recorded grassland species on an average of 39 plots/year (SE = 1.6). Following the dickcissel, in



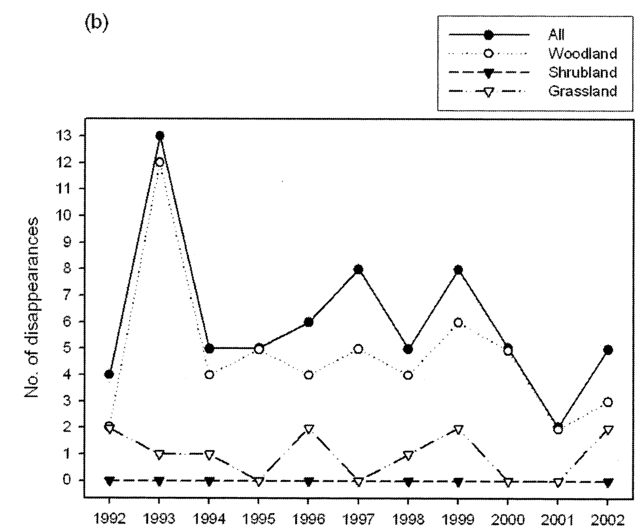


Figure 5. Appearances (a) and disappearances (b) of small landbirds at Fort Riley, 1991–2002, partitioned by breeding habitat association.

decreasing order of abundance, were brown-headed cowbird, grasshopper sparrow, eastern meadowlark, and eastern kingbird (Table 1). In contrast, the highest proportional occurrence for a shrubland or woodland bird species observed was 10 plots/year for the northern cardinal (Table 1). Coefficients of variation (CV) computed from proportional occurrences were similar between grassland (median of CV = 42.5, range = 11.4–200.0) and shrubland species (median = 51.3, range = 30.7–58.5), whereas woodland species were more variable in their presence (median = 92.1, range = 39.1–180.9; Table 1).

Based on estimates of *C*, more species declined than increased (25 vs. 15) and 7 showed no change (Table 1 and Figure 5). For the most frequently recorded shrubland and woodland species, loess plots indicated initially declining population trajectories (Figure 6a–6d) for Fort Riley. In contrast, population trajectories for most grassland species showed little change (Figure 7a–7c), except for the eastern meadowlark (Figure 7d). Based on migration assemblages, 6 of 10 resident species showed negative trends, whereas only 2 exhibited (modest) increases (Figure 8). Among short-distance migrants, 9 of 12 appeared to decline (Figure 8). For long-distance migrants, the number declining (10) nearly matched the number showing increases (11) (Figure 8).

Regionally, 27 species (57%) had *C* estimates ≥0 (Table 1). The northern flicker and loggerhead shrike showed the greatest declines; this was consistent with

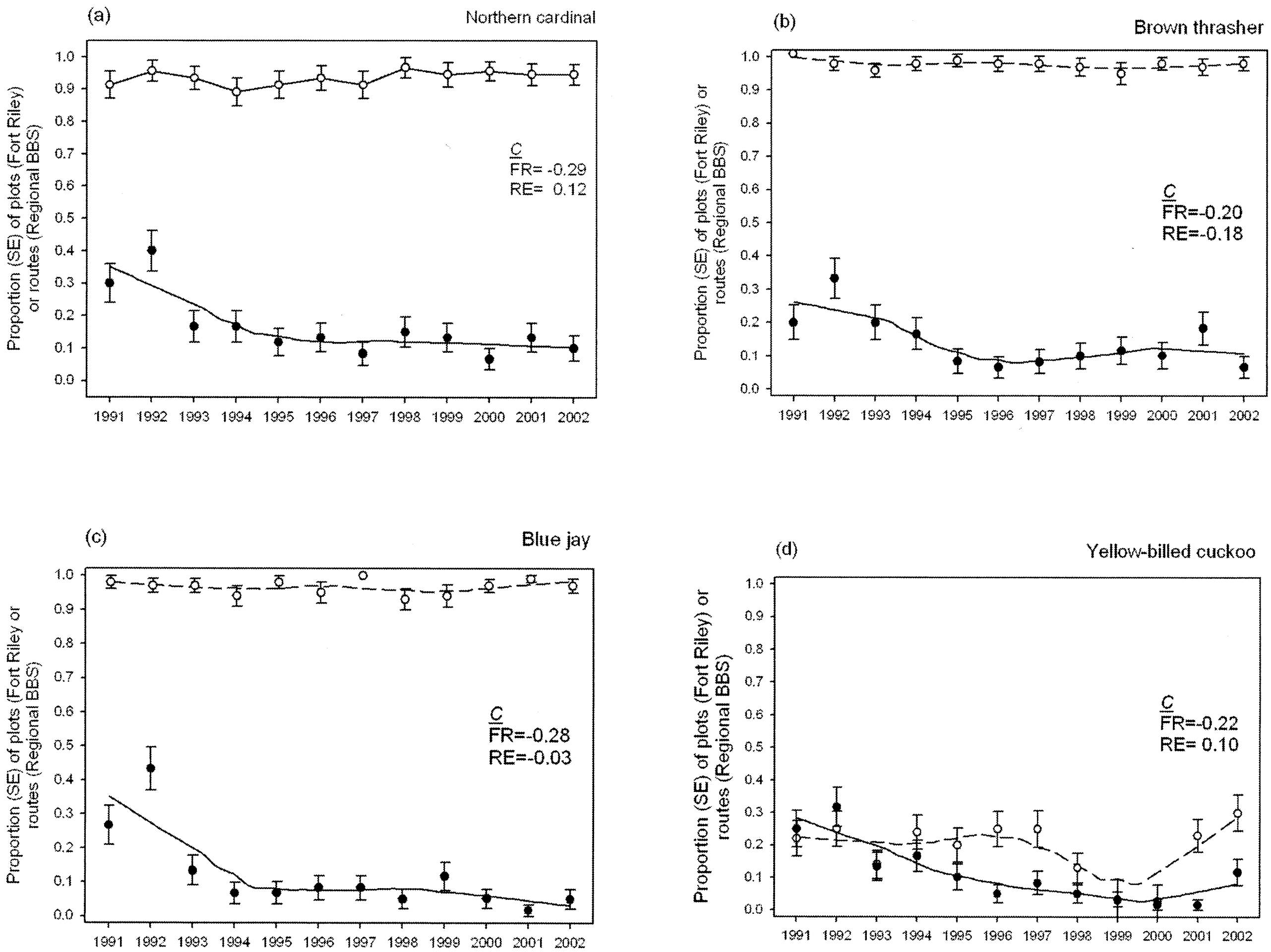


Figure 6. Index of relative difference (*C*) and loess plots illustrating population trajectories for two shrubland and two woodland species of birds at Fort Riley, Kansas, 1991–2002. Fort Riley (FR) trends data are represented by solid circles and dashed lines; regional (RE) trends data (BBS) are represented by solid triangles and dotted lines.

observations at Fort Riley. The lark sparrow, a grassland species having one of the largest negative *C* values at Fort Riley, had a large increase within the BBS region (Table 1). Among shrubland species, the northern cardinal and field sparrow had among the largest negative *C* value at Fort Riley but showed increases within the BBS region. Woodland species exhibiting the largest negative *C* values at Fort Riley compared to positive values regionally were the yellow-billed cuckoo and red-bellied woodpecker. In contrast, the Kentucky warbler was the only species regionally showing a modest decrease yet a large increase at Fort Riley. Overall, 22 of 47 (47%) species matched (either negative *C* values both locally and regionally or positive for both).

We found no large-scale changes in the coarse vegetation measures on study plots. The mean difference in the proportions of nonwoody cells/plot between 1991–1992 and 1998 was a mere 1.98% (e.g., <2% of cells changed from grassland to woodland or vice versa), indicating little (<10%) coarse-level change in vegetation. No changes were detected from the 1998 photo analysis versus the 2000 ground inspections.

Discussion

Community Measures

We found that richnesses of grassland and shrubland bird species were similar among years, whereas most woodland species were variable in their presence and generally declined. A similar pattern was found among habitat assemblages for turnover. Woodland birds underwent the greatest turnover, resulting in a net loss of species. These patterns might reflect the limited breeding habitat (~4%) in size and isolation for woodland species at Fort Riley. Moreover, Fort Riley

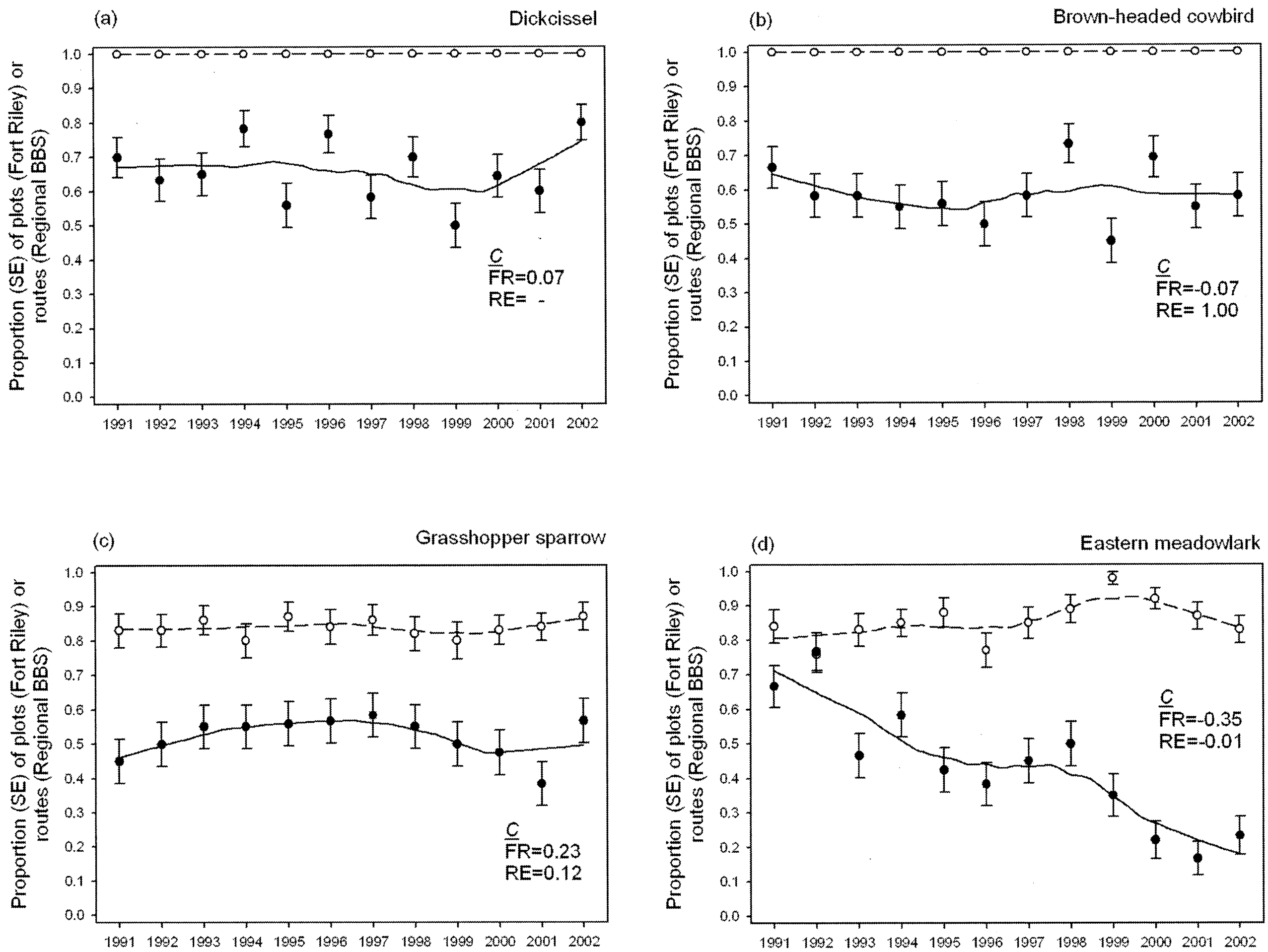


Figure 7. Index of relative difference (*C*) and loess plots illustrating population trajectories for four grassland species of birds at Fort Riley, Kansas, 1991–2002. Fort Riley (FR) trends data are represented by solid circles and dashed lines; regional (RE) trends data (BBS) are represented by open circles and dotted lines.

represents the periphery of many woodland species, and periodic disappearances and recolonizations are to be expected for this group of woodland species. Indeed, 21 of 30 woodland species were observed on ≤3 of the 60 permanent plots per year. Local populations might be too low to be self-sustaining, a characteristic of small, isolated woodlands (e.g., Donovan and others 1995; Brawn and Robinson 1996). Changes in woodland species at Fort Riley also could be impacted by troop training, which alters habitat quality. However, this is unlikely to have produced the patterns we observed because training is typically conducted in grasslands. Although the observed trends in mean turnover per plot might have been a function of sample intensity for woodland species, we suspect that these higher estimates might be more attributable to the landscape of small, isolated woodlots found at the fort.

Population Trends and Vegetation Communities

Our assessment of population change for 47 bird species using *C* estimates and loess analyses is that more species exhibited negative change (57%) than positive change (30%), with several (13%) remaining steady. Based on migration assemblages, considerably more resident and short-distance migrant species exhibited negative change than positive change. For long-distance migrants, the number with negative change essentially matched those with positive change.

Population trends of core grassland species (those observed on plots for 11–12 years), such as the dickcissel, grasshopper sparrow, and Henslow's sparrow, remained steady at Fort Riley, whereas the eastern meadowlark did not. All shrubland species were observed each year but were mixed in their *C* estimates, with three of six species exhibiting modest declines. Populations of most core woodland species declined,

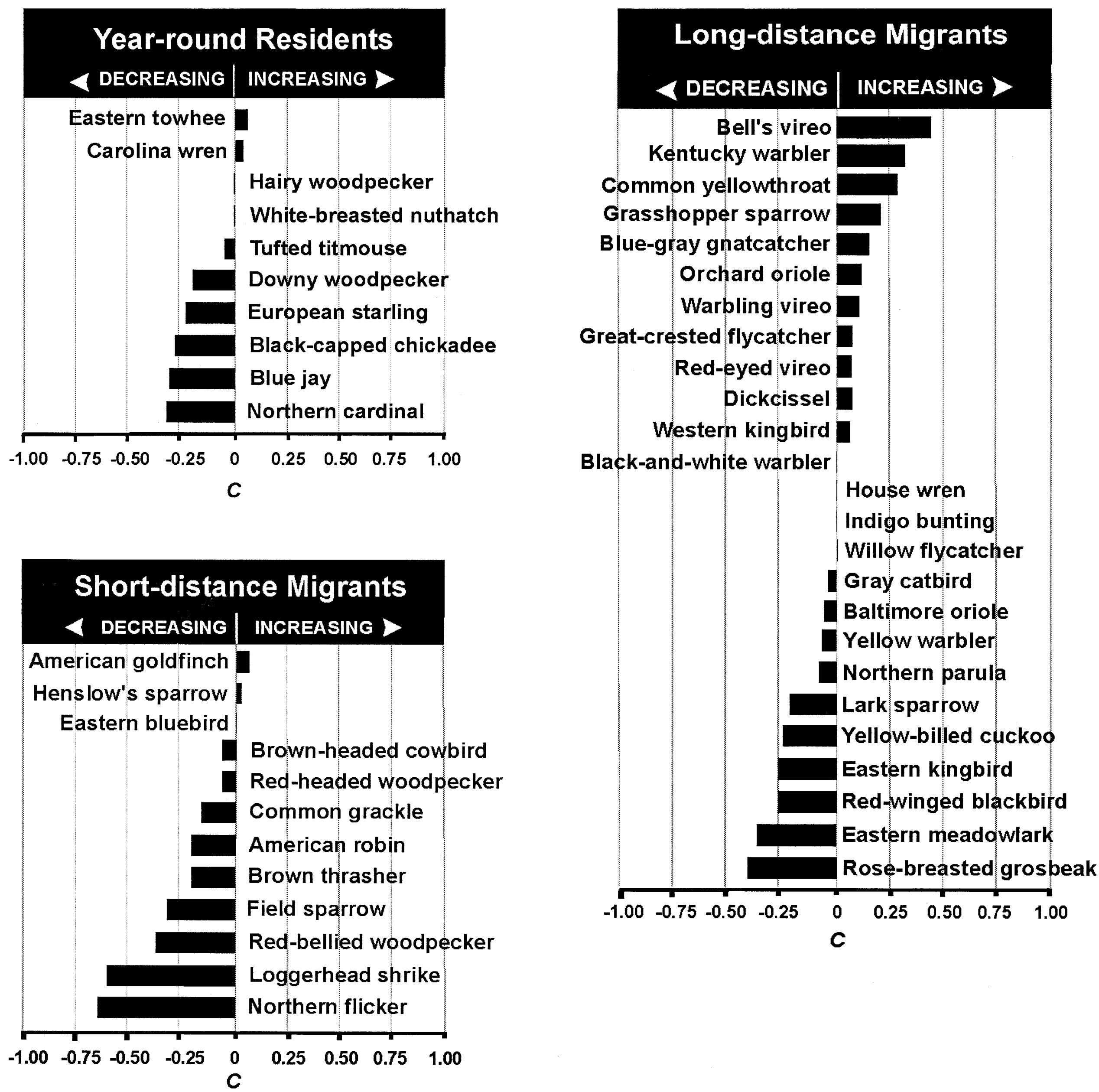


Figure 8. Index of relative difference (*C*) values of small landbird species observed ≥3 years at Fort Riley, 1991–2002, partitioned by migration status.

and these declines were observed for woodland species differing in nesting and foraging assemblages. Although 16 of the 47 species examined had relatively high PIF scores (≥20), the population changes for these 16 were mixed. For example, Bell's vireo had the third highest PIF value, yet its population increased at Fort Riley from 1991 to 2002. Thus, Fort Riley is an important site in the region because it appears to provide important breeding habitat for this species. Likewise, the same pattern was observed for other species of concern, such as the dickcissel, grasshopper sparrow, and Henslow's sparrow.

Based on differences between Fort Riley and regional *C* estimates, it appears that several species (most notably the eastern meadowlark, lark sparrow, northern cardinal, field sparrow, yellow-billed cuckoo, and red-bellied woodpecker) might be experiencing declines. These declines might be linked to disturbances or habitat changes unique to Fort Riley. This highlights the need to develop controlled experiments to determine the causal bases for such population changes. Use and interpretation of BBS data to put installation trends in better perspective must be done cautiously. Each of our plots typically represented a single habitat type, whereas BBS routes often include multiple habitat types as well as more area. The emphasis should be more on the trends, as reflected by the *C*, than the proportion of plots (the installation) or routes (BBS) in which a species is found. For example, the proportion of routes that northern cardinals have been present on BBS routes has exceeded 0.9 for all years examined compared to less than 0.4 for Fort Riley plots (Figure 6a). This comparison might lead to an interpretation that the installation management practices or land uses have a detrimental effect on northern cardinals. Although that might be the true, it might well not be the case. Thus, we recommend the use of *C* for comparing the different datasets instead of direct comparisons of proportion of plots/routes on which a species is present.

Analysis of vegetation communities indicated that habitat quantity changed little during this study. However, many species are influenced by microhabitat quality and our coarse analysis possibly missed changes that could be important for some species. For example, Henslow's sparrow and the eastern meadowlark both require significant standing residual vegetation or litter layer for breeding (Zimmerman 1988; Lanyon 1995). Both of these species prefer habitats that have not been burned (i.e., had litter reduced or residual vegetation removed) for ≥1 year. Although observed in relatively low numbers (mean of 6.5 plots/year), the Henslow's sparrow population has remained steady at Fort Riley. In contrast, eastern meadowlarks have shown one of the most precipitous decreases of all species. The decline has leveled off in recent years and we speculate that this might reflect the annual variation in the extent of annual burning. Burn data are unavailable for study plots and it remains unknown how fire might have influenced the observed patterns.

C as a Tool for Setting Monitoring Priorities

Use of *C*, in concert with loess, can provide managers with an objective tool for the selection of species that should be examined more closely at the local or regional level. *C* values serve to detect local population changes, thereby improving the objective information base from which to determine new monitoring needs and management priorities. We suggest that the conservative approach of using presence–absence for a species (vs. raw counts) and computing *C* estimates can reveal real trends over the long term. This is especially true when multiple observers (either in the same year or from year to year) are involved, which is often the case with long-term monitoring programs like LCTA.

C estimates also provide immediate feedback on the direction of change (+ or −) and the magnitude of change [−1, 1]. Although the PIF ratings (11–28 for species we evaluated) help identify species of greatest concern, *C* estimates make it easier to conceptualize the relative magnitude of change from a trend perspective. This can lead to more monitoring with different, more intensive methods to confirm and attempt to explain population changes. For example, we identified several species (eastern meadowlark and eastern kingbird) that require more in-depth study to understand the reasons for their apparent decline at Fort Riley. The results of such studies might also be useful to natural resource managers using spatially explicit ecological models to deal with land-use decisions (Dale and others 2000).

Distance sampling, which includes estimates of detection probabilities, provides a more rigorous approach to obtaining valid estimates of density estimates than raw counts (and subsequent population indices generated from the data) and should be considered in an expanded monitoring effort (Buckland and others 2001; Thompson 2002). In fact, starting in 2003, we altered our field protocol to collect the information needed for distance sampling without sacrificing the ability to compare future survey results with the existing 12-year dataset described herein. If multiple visits to plots are possible each year, it might be possible to adjust raw assessments of proportion of plots occupied for more abundant species using model approaches such as developed by MacKenzie and others (2003). In addition to estimating density, additional evaluations of reproductive success and local survival might be warranted. However, these require considerably more time and effort and are not practical for *all* species detected in annual surveys. Thus, we used *C* and loess to identify the species whose populations are of greatest concern. Just as importantly, perhaps, a large +/− value does not need to be compared to other species to identify a species at risk. Identification of declining or increasing species allows prioritization and direction of more intense monitoring and management toward those species needing the greatest conservation on a local level (Droege and others 1998).

Conclusion

There are several monitoring programs intended to assess ecosystem health via community and population measures (Karr 1991; Canterbury and others 2000; Hutto and Young 2002, 2003). Using species richness to monitor ecosystem health assumes this metric is a reliable index of habitat quality. We suggest that mean species richness/plot, rather than species richness, provides a better indicator of low to moderate changes in the bird community across Fort Riley. In other words, mean species richness/plot is a more sensitive measure of change in biological integrity than species richness because declines in mean species richness/plot will generally be observed before a species is lost areawide. Not until a dramatic decline in habitat quality is widespread or severe will total species richness measures reflect significant declines (Wilson and others 1997; Bradford and others 1998). It is plausible that species composition would change. Our estimates for turnover indicate that this was most prevalent for the woodland bird assemblages. Although species richness has been demonstrated to be a reliable indicator of forest ecosystem degradation (Canterbury and others 2000), it has limited application as a measure for grassland and rangeland ecosystems unless the disturbance is a major impact (Bradford and others 1998).

Multiple habitats exist across a large landscape at Fort Riley. It provides breeding habitat for a diversity of small landbirds. It appears that some populations of species at Fort Riley are declining. In addition to conservation concerns for neotropical migrants (Robbins and others 1989), attention is also needed for short-distance migrant and resident small landbirds as well. These assemblages have the greatest portion of species with negative change (Haney and others 2001).

Most grassland and shrubland birds appear to have adequate nesting habitat, but we currently lack data regarding reproductive success, which is vital to understanding the causes of population change. Woodland birds, with some exceptions, might lack an adequate nesting habitat to achieve stable populations at Fort Riley regardless of disturbance levels during the breeding season. This can be evaluated only with detailed studies on productivity and local survival. In addition to current monitoring of threatened and endangered species that occur at military installations, we suggest additional studies of common species to evaluate the role of military training on bird productivity and associated habitat quality.

To prioritize for species, and thus identify conservation directives, we developed an analysis approach that provides a large-scale approach based on simple computing techniques. Because other military installations have implemented similar breeding bird surveys for many years (≥10), datasets similar to Fort Riley can be evaluated using *C* and scatterplot smoothers like loess. Combined with species richness, mean species richness/plot, and turnover, installations can easily implement our approach, thereby providing an important, objective step in conservation decision-making. Long-term monitoring efforts at many national and state parks and refuges are among other management units that might benefit from implementing this methodology.

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