THE SOUTHWESTERN NATURALIST 42(2):137–147 JUNE 1997

POPULATION TRENDS OF MIGRATORY LANDBIRDS ALONG THE MIDDLE RIO GRANDE

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ABSTRACT—Using standardized mist-netting capture data from the Rio Grande Nature Center, Albuquerque, New Mexico from 1985–1994, and Breeding Bird Survey data from New Mexico and United States from 1980–1994, we compared local, regional, and national population trends of landbird species that migrate along the middle Rio Grande in fall. Evaluations of relationships among population trends, migratory distance, breeding habitat use, nest type, and foraging guilds revealed that population trends of Rio Grande migrants were relatively species-specific and that population changes were unequally distributed among habitats and life history groups. The fall banding data suggested that long-distance migrants, riparian forest migrants, canopy insectivores, and open-cup nesting species declined more over the 10-year period than other groups. Similarities between population trends from banding data and trends from Breeding Bird Survey increased as we expanded the geographic coverage of the Breeding Bird Survey data from New Mexico to the entire United States. We hypothesize that riparian habitat along the middle Rio Grande functions as a funnel that constricts habitat use during migration for species whose breeding and wintering populations are spread over broader geographic areas and that local population changes detected during mass migration may thus reflect widespread and large-scale changes.

Numerous studies, mostly from the North American Breeding Bird Survey (BBS) and from the eastern United States, have suggested that landbird species of North America, a majority of which are migratory, have experienced population declines during the last 10–15 years (Wilcove and Whitcomb, 1983; Robbins, Dawson, and Dowell, 1989; Terborgh, 1989; Askins et al., 1990; Gauthreaux, 1992; Hussell et al., 1992). For example, Rappole (1995) reported that populations of 109 species of Neotropical migrants had declined significantly within the past decade. The population status of migratory landbirds in western North America is less well-known. Region-wide data sets such as the BBS have not been analyzed adequately, and local data sets are either lacking or have not been summarized (Finch, 1991; Hejl et al., 1995). A few studies based on BBS data (Robbins et al., 1986; Sauer and Droege, 1992; Peterjohn and Sauer, 1993), counts of migrating birds (Pyle et al., 1994), and historic data (Marshall, 1988; DeSante and George, 1994) indicate that populations of some western landbird migrant species also are declining. Herkert (1995) found that several grassland bird species exhibited major population declines in the Midwest between 1966 and 1993, and DeSante and George (1994) reported that numerous bird species in the West, especially those using riparian areas for breeding, were extirpated from various states or had breeding ranges that contracted over time.

Because their habitats are restricted in distribution, total populations of western migrants are suspected to be much smaller than those of eastern species, making them particularly vulnerable to disturbance (Terborgh 1989). Factors that limit populations of eastern species may not apply to migrants that occupy western habitats of naturally small size such as narrow riparian zones along rivers and streams, or "sky islands" (e.g., isolated montane habitats of the western Mexico/United States border). In addition, travel barriers such as the Great Plains and the Rocky Mountains may limit the dispersal ability, range expansion, and elevational habitat use of western migratory landbirds (Finch, 1989). Species richness of migratory landbirds inhabiting western riparian habitats is typically higher than that in surrounding habitats such as grasslands and shrubsteppe (Carothers et al., 1974; Terborgh, 1989), yet riparian zones are undersampled by regional bird surveys. Consequently, records of long-term population trends of riparian bird species are less well known than those for their upland counterparts. For this reason, we suggest that anthropogenic modification of western riparian habitats may have had a greater impact on local and regional biological diversity than is currently reported. An estimated 95% of riparian woodland in western North America has either been degraded or eliminated in the past century by water developments, agriculture, urbanization, and domestic livestock grazing (Ohmart, 1994). In a recent assessment of endangered ecosystems in the United States, Noss et al. (1995) listed riparian ecosystems in New Mexico and Arizona as endangered. A better understanding of population trends and factors causing increases and decreases of migratory landbirds using riparian ecosystems is needed throughout the western United States before suitable on-the-ground conservation strategies can be accurately recommended or implemented.

In this study, we evaluated population trends of landbird migrants that use the middle Rio Grande as a migration corridor during fall. We compared standardized mist-netting capture data from Rio Grande Nature Center in Albuquerque, New Mexico from 1985 to 1994, to BBS data during a similar time period from New Mexico, western United States, and the entire United States. We related population trends estimated from fall migration data to life history traits such as major breeding habitat, nest type, foraging guilds, and migratory status to isolate traits or behaviors that could potentially be linked to factors causing population changes.

METHODS—*Rio Grande Banding Data*—The Rio Grande Nature Center (RGNC) is located along the middle Rio Grande in Albuquerque, Bernalillo County, New Mexico (35°07′N, 106°41′W). Rio Grande Bird Research, Inc. established a long-term banding station at RGNC and has mist-netted landbirds during fall migration since 1979. The study area was established in a riparian habitat. The overstory vegetation is dominated by Rio Grande cottonwood (*Populus fremontii* var. *wislizeni*) and also includes Russian olive (*Elaeagnus angustiflia*), New Mexico olive (*Forestiera neomexicana*), Goodding willow (*Salix gooddingii*), false indigo (*Amorpha fruticosa*), seepwillow (*Baccharis glutinosa*), coyote willow (*Salix exigua*) and Salt Cedar (*Tamarix* spp.). The dominant understory includes skunkbush (*Rhus trilobata*), Arizona grape (*Vitis arizonica*), alkali sacaton (*Sporobolus airoides*), sand dropseed (*S. cryptandrus*), rabbitbrush (*Chrysothamnus nauseosus* var. *bigelovii*), tumbleweed (*Salsola kali*), and *Kochia* sp. The area was excluded from public access, and habitat disturbance or modification during the study was not evident (Cox, 1994).

Twenty mist nests (12 m × 2.6 m with 30 mm or 36 mm mesh) were used to capture and recapture landbirds from early August through mid-November each year. Nets remained at the same locations through the study period and were opened about 15 minutes before local sunrise and remained up for approximately six hours each banding day. Nets were checked every 20 to 30 minutes. Nets were closed during rain, snowfall, and other excessive weather conditions (i.e., temperature >38°C or winds >40 km/h). Species, age, and sex identification were based on the North American Bird Banding Manual (U.S. Fish and Wildlife Service and Canadian Wildlife Service, 1984), Identification Guide to North American Passerines (Pyle et al., 1987), and various field guides. Each bird was banded with a U.S. Fish and Wildlife Service aluminum leg band (see Yong et al., 1995 for more information).

The mist nets at RGNC were operated by volunteers during weekends in fall. We consider weekend mist-netting sampling to be valid for monitoring population changes of migratory birds in studies that emphasize a population index instead of maximum numbers of captures (Dunn and Hussell, 1995) and when stopover biology, including mass gain and stopover length, is extraneous to a study's objectives. The number of total captures from weekends is often correlated with that from weekdays of the same week (Yong and Finch pers. obs.). We treated yearly weekend mist-netting efforts as temporally stratified repeated samples. We recognized that the daily capture total of each species could be affected by factors such as weather conditions and species-specific seasonal migration patterns, but considered these factors to be of stochastic influence on the annual totals and population trends through the ten-year period. We adjusted net-hour variation among years by calculating yearly indices for each species. We divided the total number of captured individuals by the total number of net hours of a given year, and then multiplied the resulting numbers by 100. This produced a yearly estimate of birds/100 net-hours of banding operation (Titus and Mosher, 1982; Hussell, 1985; Bednarz et al., 1990).

To examine population trends, we performed regression analyses with year as the predictor variable and birds captured/100 net-hours of each species as dependent variables. The slopes of the regression equations were used as indexes of population trends. Only common species with total capture numbers ≥30 were used in trend analyses. We defined trends as significant at $P < 0.1$. We chose this alpha level because Type I error is less risky than Type II error in detecting population declines of migratory birds. If population changes can be detected early due to initial warnings from trend data, the species has a greater chance of population recovery through close monitoring and prompt conservation actions. We used log-likelihood ratio (G) tests to examine the hypotheses that population trends based on netting data were independent of migration status, breeding habitat, nest type, and foraging guilds. We classified trends as positive or negative based on direction of slopes and included all species, not just those having significant trends (Robbins, Dawson, and Dowell, 1989; Hagan et al., 1992; DeSante and George, 1994; Pyle et al., 1994; Herkert, 1995).

Breeding Survey Data—We retrieved BBS trend data between 1980 and 1994 from National Biological Service's database. These trends were estimated using the route-regression method (Geissler and Sauer, 1990). Regional trends were estimated as a weighted average of trends on individual routes. Route trends were estimated using the estimating equations estimator (Link and Sauer, 1994), in which a multiplicative trend was estimated. Observer effects were incorporated into the model to prevent bias associated with increases in observer quality over time (Sauer et al., 1994).

Classification of Life-history Groups—We followed Robbins, Sauer, et al. (1989) and Hussell et al. (1992) in classifying species as tropical or temperate migrants in accordance with winter distributions (AOU, 1983; Root, 1988). Migratory species wintering mainly north of 25°N were classified as temperate and all others were tropical migrants. We assigned species to the following three breeding habitat use categories: forest/woodland, shrub/scrub/grassland, and other habitats (Hubbard, 1978; AOU, 1983; Ehrlich et al., 1988). Foraging guilds were classified as foliage insect gleaners, ground insect gleaners, other insectivores, granivores, and others (AOU, 1983; Ehrlich et al., 1988). Nest types were classified as cavity nesters, canopy open-cup nesters, low vegetation open-cup nesters, and others.

RESULTS—*Population Status from Netting Data*—During the ten-year netting operation, 15,625 birds of 108 species (Mean = 1562 birds/yr, *SE* = 194) were captured during a total of 30,431 net-hr (Mean = 3,043 net-hr/yr, *SE* = 155). Forty-one species had sample sizes ≥30 and accounted for 15,302 (98%) of the total captures (Table 1). These 41 species were all migratory, including 22 species (53%) of Neotropical migrants and 19 species (47%) of temperate migrants. Population trend analyses were performed on these 41 species.

Population trends for 22 of 41 species (54%) had negative slopes for the 10-year study period, of which three species—western tanager ($r = -0.67$, $P = 0.06$), house finch ($r = -0.59$, $P = 0.07$), and spotted towhee ($r = -0.54$, $P = 0.01$)—showed significant population declines (Table 1). Trends of solitary vireo ($r = -0.53$) and pine siskin ($r = -0.56$) populations tended toward significant decreases ($P = 0.14$ and 0.15, respectively). Additionally, brown creeper (*Certhia americana*) and yellow-billed cuckoo (*Coccyzus americanus*) were captured less often as the study progressed, but these species were not included in trend analyses because of small sample sizes.

Trends for 19 of 41 species (46%) had positive slopes during the study period. Significant increases were detected in seven species: northern flicker ($r = 0.70$, $P = 0.08$), Bewick's wren ($r = 0.65$, $P = 0.04$), black-capped chickadee ($r = 0.77$, $P = 0.01$), hermit thrush ($r = 0.66$, $P = 0.04$), Wilson's warbler ($r = 0.70$, $P = 0.02$), white-crowned sparrow ($r = 0.64$, $P = 0.05$), and American goldfinch ($r = 0.53$, $P = 0.10$).

Trends in Relation to Life Histories—Species with negative and positive trends based on banding data were not equally distributed among habitat and life history classes. About two-thirds of forest breeding species showed, or tended toward, population declines over the study period, while only one-third of the shrub and grassland breeding species showed negative slopes ($G = 6.73$, $d.f. = 2$, $P = 0.03$, Fig. 1). These trends were more obvious among species using riparian habitat for nesting. Eighty-two percent of riparian-forest breeding species tended toward population declines, while 71% of the riparian-shrub breeding species showed positive slopes (Table 1). Two species (willow flycatcher and red-winged blackbird) that nest close to water showed negative slopes.

While 64% of the Neotropical migrants had negative population slopes, 58% of the short-distance migrants had positive slopes. Contingency analysis suggested that slope direction was related to foraging guilds ($G = 7.24$, $d.f. = 3$, $P = 0.06$). While more ground insectivores and granivores had or tended toward positive slopes, more than 75% of the foliage insectivores and other insectivorous species exhibited negative slopes (Fig. 2). Many of these foliage insectivores are Neotropical migrants such as vireos, warblers, and tanagers.

Proportionately, the open-cup canopy-nesting group had more species with negative slopes (western wood-pewee, American robin, western tanager, solitary and warbling vireos, Virginia's warbler, black-headed grosbeak, house finch, and pine siskin) than positive slopes (ruby-crowned kinglet and yellow-rumped warbler), while the pattern was reversed in open-cup ground or low vegetation nesters

TABLE 1—Life history characteristic and population trends of landbirds captured at Rio Grande Nature Center in Albuquerque, New Mexico from 1985 to 1994.

				Guild[d]		Banding[e]		BBS trend[f]		
Common name	Scientific name[a]	Breeding[b] habitat	Migratory[c] status	Nesting	Foraging	*n*	Trend	NM	Western US	US
Picidae										
Northern flicker	*Colaptes auratus*	F	B	O	GI	31	+*	+	+*	+*
Tyrannidae										
Western wood-pewee	*Contopus sordidulus*	F	A	TC	OI	43	−	+	−*	−*
Willow flycatcher	*Empidonax traillii*	O	A	SC	OI	59	−			
Dusky flycatcher	*Empidonax oberholseri*	F	A	SC	OI	98	−		−	+
Paridae										
Mountain chickadee	*Parus gambelli*	F	B	CA	FI	40	−	−	−*	−*
Black-capped chickadee	*Parus atricapillus*	F	B	CA	FI	51	+*		−*	+
Troglodytidae										
Bewick's wren	*Thryomanes bewickii*	F	B	CA	GI	51	+*	+*	−*	−
House wren	*Troglodytes aedon*	S	A	CA	GI	94	−		+*	+*
Muscicapidae										
Ruby-crowned kinglet	*Regulus calendula*	F	B	O	FI	125	+		−	+
Hermit thrush	*Catharus guttatus*	F	B	SC	GI	271	+*		+	+*
American robin	*Turdus migratorius*	O	B	TC	GI	170	−	+		+
Vireonidae										
Solitary vireo	*Vireo solitarius*	F	A	TC	FI	32	−	+*	+*	+*
Warbling vireo	*Vireo gilvus*	F	A	TC	FI	38	−		+*	−
Emberizidae										
Parulinae										
Orange-crowned warbler	*Vermivora celata*	F	A	SC	FI	433	−		+*	+
Virginia's warbler	*Vermivora virginiae*	F	A	SC	GI	129	−		+*	+*
Yellow warbler	*Dendroica petechia*	S	A	SC	FI	190	−			+*
Yellow-rumped warbler	*Dendroica coronata*	F	B	TC	FI	610	+		−	−*
MacGillivray's warbler	*Oporornis tolmiei*	F	A	SC	FI	356	−	−	−	+
Wilson's warbler	*Wilsonia pusilla*	S	A	SC	FI	961	+*		−*	−*
Thraupinae										
Western tanager	*Piranga ludoviciana*	F	A	TC	FI	153	−*	+	+*	+
Cardinalinae										
Black-headed grosbeak	*Pheucticus melanocephalus*	F	A	TC	FI	190	−	+	+*	+*
Blue grosbeak	*Guiraca caerulea*	S	A	SC	GI	292	+	+*	+*	+*
Lazuli bunting	*Passerina amoena*	S	A	SC	GI	451	+		+	+*
Emberizinae										
Green-tailed towhee	*Pipilo chlorurus*	F	A	SC	GI	141	+		+	+
Spotted towhee	*Pipilo maculatus*	F	B	SC	GI	126	−*	+	+*	−*
Chipping sparrow	*Spizella passerina*	F	A	TC	GI	4,221	+	−	−	+*
Clay-colored sparrow	*Spizella pallida*	S	A	SC	GI	86	+		−	+*
Brewer's sparrow	*Spizella breweri*	S	A	SC	GI	202	−		−*	−*
Vesper sparrow	*Pooecetes gramineus*	S	B	SC	GI	110	+	−	+*	−
Lark sparrow	*Chondestes grammacus*	S	A	SC	SE	426	+	−	−	−
Savannah sparrow	*Passerculus sandwichensis*	S	B	SC	GI	109	−		+	−
Song sparrow	*Melospiza melodia*	S	B	SC	GI	261	+		−*	+*
Lincoln's sparrow	*Melospiza lincolnii*	S	A	SC	GI	284	+		+*	+

TABLE 1—Continued.

Common name	Scientific name[a]	Breeding[b] habitat	Migratory[c] status	Guild[d] Nesting	Foraging	Banding[e] n	Trend	BBS trend[f] NM	Western US	US
White-crowned sparrow	*Zonotrichia leucophrys*	S	B	SC	GI	1,089	+*		+	−
Dark-eyed junco	*Junco hyemalis*	F	A	SC	SE	915	−		−*	−*
Icterinae										
Red-winged blackbird	*Agelaius phoeniceus*	O	B	O	GI	30	−	−	−	−*
Northern oriole	*Icterus galbula*	F	A	O	FI	88	−	+	+	+
Fringillidae										
House finch	*Carpodacus mexicanus*	S	B	TC	SE	1,657	−*	+	+	+*
Pine siskin	*Carduelis pinus*	F	B	O	SE	266	−		−	−
Lesser goldfinch	*Carduelis psaltria*	S	B	TC	SE	81	+	+	−	−
American goldfinch	*Carduelis tristis*	F	B	SC	SE	342	+*		−	+*

[a] Based on the AOU Check-list of North American Birds (1983) and its supplements.

[b] Migratory species wintering mainly north of 25°N are classified as temperate migrants (A) and all others are tropical migrants (B).

[c] Major breeding habitats: F = forest, S = shrub, scrub, and grassland, and O = other habitats (Hubbard, 1978; AOU, 1983; Ehrlich et al., 1988).

[d] Nest type: CA = cavity nesters, TC = canopy open-cup nesters, SC = low vegetation open-cup nesters, and O = other nest types. Foraging guilds: SE = granivores, GI = ground insect gleaners, FI = foliage insect gleaners, OI = other insectivores, and O = other foraging types (AOU, 1983; Ehrlich et al., 1988).

[e] Banding data from Rio Grande Nature Center in Albuquerque, New Mexico between 1985 and 1994. *n* = birds captured. * $P < 0.1$.

[f] Population trends based on Breeding Bird Survey (BBS) between 1980 and 1994: NM, WE, and US = New Mexico, western United States, and entire United States, respectively. * $P < 0.05$.

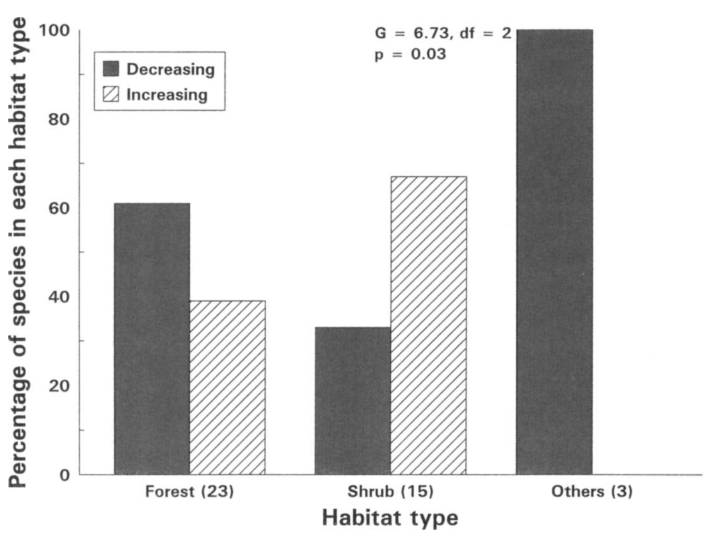

FIG. 1—Relationship between population trends and breeding habitat types. Forest = forest/woodland and Shrub = shrub/scrub/grassland. The numbers in paretheses are the total species in each habitat type.

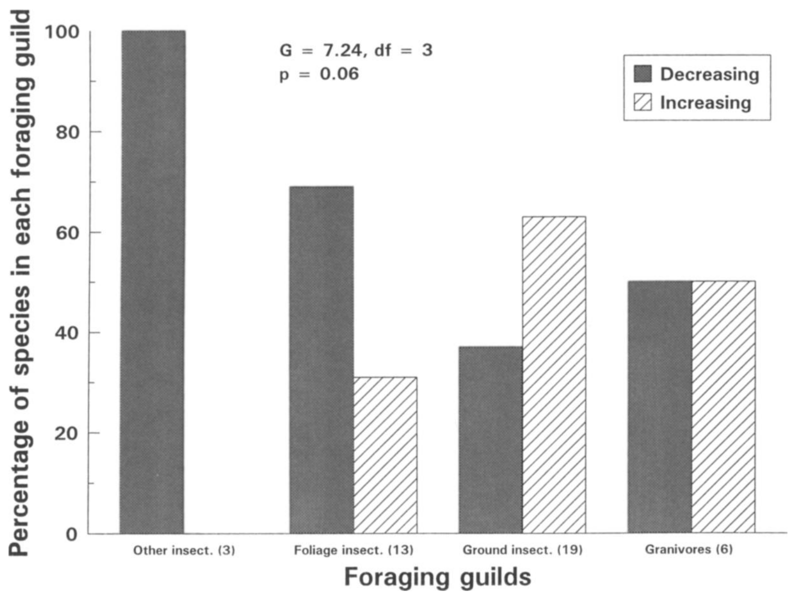

FIG. 2—Relationships between population trends and foraging guilds: foliage insectivores, ground insectivores, other insectivores, and granivores. The numbers in parentheses are the total species of each foraging guild.

(Table 1). However, about 50% (22) of the total species (41) were ground or low-vegetation nesters. Thus, the absolute number of species with negative trends in this group was higher than that in open-cup canopy nesters.

Comparisons between Netting Data and BBS Trends—Trends from New Mexico BBS data were available for only 18 of the 41 species for which we had estimated trends from Rio Grande netting data. BBS data from the western United States and the entire United States had trends available for 38 and 40 of the 41 species, respectively (Table 1). When we enlarged the geographic scale of the BBS trends from New Mexico to western United States and then to the entire United States, the number of species that had consistent slopes (same directions) between netting data and BBS data increased from 8 (39%) to 16 (42%), then to 21 (53%), respectively (Fig. 3).

Of the banding data and New Mexico BBS data, slopes were consistently negative for mountain chickadee, MacGillivray's warbler, red-winged blackbird and were consistently positive for northern flicker, Bewick's wren, blue grosbeak, and lesser goldfinch (Table 1). Among the 40 species for which both BBS trends for the entire United States and trends from netting data were available, slopes of nine species (23%) were consistently negative between data sets (Table 1). Five of these species nest mostly in coniferous or deciduous forests (dark-eyed junco, mountain chickadee, warbling vireo, pine siskin, and western wood-pewee); two species breed primarily in aridland habitats (Brewer's sparrow and savannah sparrow); one inhabits shrubby thickets (spotted towhee); and one species breeds in marshes (red-winged blackbird). Another 12 species (30%) showed consistent positive slopes (Table 1). Of these, at least seven nest in shrubby habitats (American goldfinch, blue grosbeak, green-tailed towhee, lazuli bunting, Lincoln's sparrow, song sparrow, and white-crowned sparrow); six species, some of which are repeated, are associated with forest clearings, edges, and second growth (American goldfinch, black-capped chickadee, chipping sparrow, lazuli bunting, white-crowned sparrow, and northern flicker); and hermit thrush and ruby-crowned kinglet inhabit continuous forest.

DISCUSSION—*Population Status of Migratory Landbirds*—The results of this study suggest that population trends of migratory landbird species estimated from middle Rio Grande netting data show patterns of change related to both habitat and life history classifications. Over the 10-year duration of this study, captures of Neotropical long-distance migrants, forest species, canopy insectivores, and open-cup canopy nesting species were more likely to decrease, while short-distance migrants, shrubland species, ground insectivores and granivores, and shrub-nesting species were more likely to increase or remain stable. At the general level of forest vs. non-forest habitats, these patterns are compatible with findings in the eastern United States (Whitcomb et al., 1981; Robbins et al., 1986, 1989a; Askins 1993)

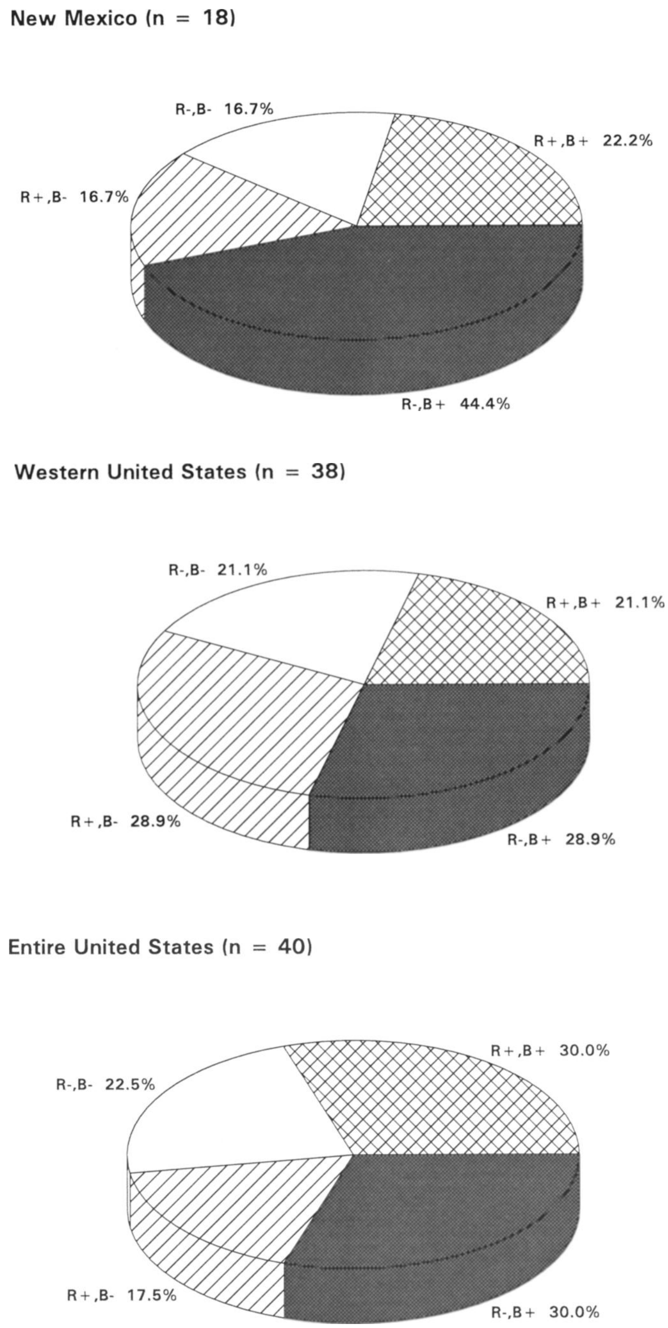

FIG. 3—Comparisons between trends estimated from netting data collected at Rio Grande Nature Center in Albuquerque, New Mexico and trends from Breeding Bird Survey for New Mexico. R = population trends from Rio Grande Nature Center banding data, B = population trends from BBS, and "+" and "−" indicate positive and negative trends, respectively.

and some available studies in the western United States (Robbins et al., 1986; Sauer and Droege, 1992; Peterjohn and Sauer, 1993). Nevertheless, our study results emphasized that trend differences were most pronounced among species that nested in riparian trees vs riparian shrubs, a finding apparently unique to our study of western birds. The fall population changes we detected were more likely to be related to conditions at breeding sites rather than direct responses to habitat changes at our study site because: (1) during our study period, the habitat at our particular study site remained unaltered except for vegetation aging, (2) a parallel study by Cox (1994) at the same banding station did not detect any significant effects of local disturbances on mist-netting capture rates, (3) trends among species using similar habitats or within the same life-history guilds were different (suggesting spatial differences among breeding sites), and (4) for about half of the species analyzed, our netting data produced trends that were similar to those from the BBS data for a much larger geographic area (also see Hagan et al., 1992).

Fifty percent of the 41 species captured in this study were ground or shrub nesters, while only 23% were canopy nesters. Although proportionately fewer ground or shrub nesters had negative trends, the absolute number having declining trends was higher than that of canopy-nesting species. To conserve all groups of concern, we recommend that conservation priorities be fine-tuned by carefully exploring relationships within each habitat/behavioral class. To determine conservation priorities, additional factors such as rarity or endangerment of each species and habitat, species and habitat vulnerability to disturbance and fragmentation, and severity of population and habitat declines should be considered. Nevertheless, linking trend patterns to habitats and life histories produces a broad habitat conservation approach for multiple species that may be more cost-effective and less short-sighted and species-biased than traditional single-species approaches (Block et al., 1995).

Several eastern studies (e.g., Whitcomb et al., 1981; Wilcove, 1985; Terborgh, 1989) indicate that forest fragmentation and urbanization may result in increased nest predator abundance or nest detection and destruction rates by predators, resulting in lowered breeding success, recruitment rates, and populations of targeted species. Ground- or low-vegetation nesting species are assumed to be more vulnerable to nest predation in fragmented forests than canopy nesters because low nests may be more visible and accessible to predators. Studies in the West and Midwest (e.g. Martin, 1993; Herkert, 1995), however, have shown that low-nesting species do not have greater nest predation rates than high-nesting species. If we assume that capture trends during fall migration reflect changes in breeding productivity over time, then our netting results agree with the idea that canopy-nesting species are as vulnerable to nest failure as species nesting in other substrates; that is, proportionately more canopy open-nest species showed population declines in our study than did cavity, shrub, or ground-nesting birds. Many canopy-nesting species are also foliage insectivores that glean or flycatch insects in forest canopies. These include long-distance migrants with negative trends such as solitary vireo, warbling vireo, western wood-pewee, and western tanager. If populations of more canopy species have declined over the past 10 years while more shrub- and ground-nesting species have increased as suggested by our migration data, then actual amounts of forest and shrub habitats, or essential components therein (e.g., food, nest sites, foraging substrate), may have shifted (owing to factors such as succession, habitat manipulations, alien plant invasions) over the study period, at least for those populations we sampled. Alternatively, populations of nest predators may have fluctuated, causing fluctuations in breeding productivity and consequently migration trends of bird species.

Results from this study highlight trends in several riparian and wetland species, suggesting that populations of riparian shrubland species such as blue grosbeak, American goldfinch, song sparrow, and Lincoln's sparrow are doing well, while marsh and backwater species (e.g., red-winged blackbird and willow flycatcher) may be losing ground. In western riparian zones where exotic shrub species such as salt cedar and Russian olive have increased, this understory invasion could explain some of the shifts in populations of riparian bird species. Loss of flooding which produces backwater habitat and loss of riparian wetlands to draining, drying, and channelization may explain decreases in wetland birds. Native willow replacement by exotic woody plants and loss of still backwaters with dense surrounding thickets may partially explain the disappearance of breeding populations of the endangered southwestern willow flycatcher (*E. t. extimus*), from most southwestern riparian woodlands.

To a certain degree, riparian habitat along the middle Rio Grande may represent a bottleneck during migration because many North American species and individuals funnel into these limited habitats, especially in the fall. Not only were New Mexico breeding species captured during their southward flights down the Rio Grande, but numerous species breeding in other western states and even several eastern species also were caught. The high species richness and population density of migratory landbirds in the Rio Grande Valley and similarities between trends from banding data and national BBS data for several species support the hypothesis that bird populations breeding in numerous habitats and states could potentially be influenced by vegetation changes along the Rio Grande. Patterns of bird migration and habitat use in other southwestern riparian ecosystems are consistent with our views (Hubbard, 1971; Carothers et al., 1974; Ohmart and Anderson, 1982). During fall migration, a large portion of Rio Grande migrants are young, hatching-year birds thought to be especially vulnerable to navigational mistakes, starvation, and predation on their first journey south to the wintering grounds. We argue that disturbance (e.g., burning, bridges, recreation, urbanization, and grazing) and habitat structural and compositional changes of the Rio Grande bosque will increase the probability that migration for some species will be altered or disrupted, and that such changes will affect not only local New Mexico birds but also populations from a much wider geographic region.

Comparisons between BBS and Netting Data—The BBS provides important information on the population status of many migratory songbird species, and in many parts of the country it is the only available long-term data set that can be used to generate population trends. Dunn and Hussell (1995) found a significant correlation between trends in numbers banded at a station in southern Michigan and Michigan BBS trends in 11 migratory songbird species. They suggested that migration counts and captures could be especially valuable in tracking population trends in species whose breeding density is very low or that do not breed primarily in habitats sampled by roadside surveys such as BBS. In the present study, population trends estimated from BBS data in New Mexico were only available for a small number of Rio Grande migrant species. Several factors could contribute to this low overlap in data sets. First, most species captured during fall migration breed outside New Mexico or have a limited breeding range within New Mexico, such as American goldfinch, clay-colored sparrow, dark-eyed junco, orange-crowned warbler, savannah sparrow, and white-crowned sparrow (Hubbard, 1978). These abundant migrants use the middle Rio Grande as a major movement corridor during spring and fall but are absent or uncommon at other times of the year. Thus, BBS routes in New Mexico are not likely to sample them in numbers sufficient enough to estimate trends (Williams pers. comm.).

Second, the randomized placement of BBS routes depends on the availability of secondary roads. While this design can be readily implemented in most parts of the eastern United States, it poses problems in many parts of New Mexico where roads are few and distributed nonrandomly. Riparian zones are known to be undersampled by BBS because most roads travel across rather than within floodplains. For this reason, BBS data for many common species breeding in riparian or wetland habitats of New Mexico, such as house wren, Wilson's warbler, and yellow warbler, are not sufficient to generate population trends. Inconsistencies in how many and which BBS routes are sampled each year further reduces the probability of accurately tracking population changes in New Mexico. Results from this study suggest that as BBS sample sizes and geographic scales increased, similarities between trends from BBS and fall banding data increased.

Third, BBS data show that population trends may be very different for the same species in different regions. If the Rio Grande banding station is sampling a species from diverse and numerous breeding populations, some of which are declining and some increasing, then the cumulative data may mask clear trends. Hagan et al. (1992) found that regional trends in eastern Neotropical migrants did not mirror overall population declines and indicated that the value of population trends estimated from banding data is limited to the general region of sampling. In contrast, we detected similarities between local and continental populations for 53% of the species we analyzed. The temporal overlap of these two data sets was not perfect (1985–1994 banding data, 1980–1994 BBS data), which may partially explain dissimilar trends between the two data sets for some species.

Inconsistencies between the study by Hagan et al. (1992) and our study could result from differences in migration pattern and availability of *en route* habitats between the western and eastern United States. In the eastern United States, *en route* habitats are more abundant and widely distributed, while migration habitats in the West are concentrated over major waterways and mountains. Lincoln (1979) suggested that the number of migration routes in the West was limited, which might result in high fidelity of flight route used by individuals and populations. This could lead to lower variation in numbers of migrants captured among years in the West than in the East and higher correlations between population trends estimated from migration netting data and large-scale trends observed on the breeding grounds.

We thank Rio Grande Bird Research, Inc. for providing the banding data for our analyses and its volunteer members who spent long hours collecting migratory bird data during the last 17 years at the Rio Grande Nature Center, Albuquerque, New Mexico. We are grateful to the staff at the Rio Grande Nature Center for access to banding sites and assistance with the banding program. We especially acknowledge J. R. Sauer, USGS Biological Resources Division, for supplying BBS trend data, and the thousands of BBS volunteer observers, recorders and coordinators whose dedicated work has provided the data necessary to estimate breeding bird population trends. J. P. Hubbard and S. O. Williams III supplied information on the Breeding Bird Survey in New Mexico. G. R. Bodner, D. Hawksworth, and G. R. Paglia assisted with data collection and entry. We thank S. O. Williams, F. R. Moore, M. Means, J. Hagan, and an anonymous reviewer for offering helpful comments and suggestions on the manuscript.

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