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Effects of Recreational Activity and Livestock Grazing on Habitat Use by Breeding Birds in Cottonwood Forests Along the South Fork Snake River

by
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EFFECTS OF RECREATIONAL ACTIVITY AND LIVESTOCK GRAZING ON HABITAT USE BY BREEDING BIRDS IN COTTONWOOD FORESTS ALONG THE SOUTH FORK SNAKE RIVER

by

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PREFACE

The study reported here was part of a larger research effort to evaluate the influence of surrounding landscapes and local land-use practices on habitat relationships of breeding birds in cottonwood riparian forests (Saab 1996). The intent of these studies was to provide information to managers on habitat features that are necessary for the long-term persistence of small landbirds breeding in cottonwood forests. Data presented in this report are based on the distribution and abundance of breeding birds in relation to local land-use practices of livestock grazing and recreational activities, while a report is forthcoming that examines the effects of these practices on nesting success of small landbirds. The influence of surrounding landscapes on habitat use by breeding birds is reported in Saab (1999). In that paper, the relative importance of several spatial scales to habitat selection by birds is examined, including the landscape scale (composition and patterning of surrounding vegetation types and land uses), macrohabitat (cottonwood forest patch characteristics), and microhabitat (local vegetation characteristics). Management implications from the spatial scale paper (Saab 1999) are included in the last section of this report to provide one reference for managers.

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ABSTRACT

More than any other habitat in western North America, arid-land riparian woodlands are centers of high diversity and abundance of birds. Because these habitats are fragmented and limited in distribution, western riparian birds might be particularly vulnerable to human-caused disturbances. During a four-year study, I examined the influences of land-use practices in relation to cottonwood forest-patch dynamics on bird community and vegetative characteristics in southeastern Idaho. Patterns in bird community characteristics of 34 species, relative abundances of individual species and nest guilds, and vegetation structure were compared among three land uses (areas managed for livestock grazing [grazed], areas managed as campgrounds [recreation], and areas not managed for grazing or recreation but for riparian and wildlife habitat values [unmanaged]), and three patch-size classes (small [<1-3 ha], medium [>3-10 ha], and large [>10-204 ha]). Overall species richness, diversity, evenness, and turnover remained fairly constant among all land uses. On average, species numbers and relative abundance appeared to be most reduced by recreational activities except in large patches managed for recreation. Few differences existed between grazed and unmanaged sites in overall mean number of species or mean number of individuals per survey visit. However, distribution and abundance of individual species, and species grouped by nest layer and nest type, varied significantly among land-use activities and patch sizes. Vegetation structural characteristics within the ground, shrub, and canopy layers were positively correlated with abundance of birds nesting in those layers. Ground-nesting species (Veery and Fox Sparrow) were most susceptible to disturbances created by livestock grazing and were also most sensitive to fragmentation of riparian habitats. Canopy nesters, including cavity-nesting species, responded positively in grazed habitats, while shrub-nesting species tended to decrease with grazing and recreational activities. Significant results of Poisson regression, for 17 of 30 species analyzed, suggested differential effects of land use, patch size, and/or the interaction between the two effects. Relative abundances of 11 species decreased with either grazing or recreation, whereas six species increased with these same activities. Five species (Gray Catbird, Veery, Yellow Warbler, Black-headed Grosbeak, and American Goldfinch) were unaffected by patch size in unmanaged areas but showed significant area effects (increases in probability of occurrence with cottonwood forest area) in grazed and/or recreation sites. Results of my study suggest that conservation of large patches is particularly important where riparian forests are managed for grazing and recreation. Apparently, some species need larger patches of breeding habitat in areas with these disturbances. In addition to evaluating the effects of local land-use practices on habitat relationships of breeding birds, I examined the importance of landscape patterns to habitat use by birds. Among three spatial scales (landscape, macrohabitat, and microhabitat), landscape features were the most important and frequent predictors of distribution and abundance for most bird species and for predicting high species richness of native avifauna. Thus, surrounding landscape features should be a primary consideration for managing riparian habitats and selecting riparian reserve areas.

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EXECUTIVE SUMMARY

More than any other habitat in western North America, arid land riparian forests are centers of high diversity and abundance of birds. Because these habitats are naturally fragmented and limited in distribution, western riparian birds might be particularly vulnerable to human-caused disturbances. During a four-year (1991-1994) research project, we examined patterns of habitat use by breeding birds in cottonwood riparian forests in relation to land-use practices, forest-patch dynamics, and spatial scale. Bird distribution and abundance, and vegetation characteristics were quantified on 57 cottonwood forest patches, ranging in size from 0.40 ha - 205 ha, along 100 km of the South Fork of the Snake River in southeastern Idaho. The goal of this work was to provide information to managers on habitat features that are necessary for the long-term persistence of migratory landbirds breeding in cottonwood forests.

Factors potentially influencing habitat selection in relation to livestock grazing, recreational campgrounds, and forest-patch dynamics were analyzed in discrete categories by land use (grazed, recreation, and unmanaged), and patch size (small [< 1- 3 ha], medium [> 3 - 10 ha], and large [> 10 ha]). On average, species richness and relative abundance were most reduced in recreational campgrounds, except in large patches managed for recreation. Ground-nesting Neotropical migrants were most susceptible to disturbances created by livestock grazing and were also most sensitive to fragmentation of riparian habitats. Five species were unaffected by patch size in unmanaged areas but showed but showed significant area effects (increases in probability of occurrence with increases in forest area) in grazed and/or recreation sites. These results suggest that some species may need larger patches of breeding habitat in areas with these disturbances.

To examine the importance of spatial scale to habitat use, a hierarchical approach was used at three scales: microhabitat (local vegetation characteristics), macrohabitat (features of cottonwood forest patches), and landscape (composition and patterning of surrounding vegetation types and land uses). The surrounding landscape changed from a valley surrounded by mountains on the upstream end of the study area, a narrow canyon adjacent to upland natural vegetation in the middle section, to a wide, flat floodplain dominated by agriculture on the downstream end. The best predictors of high species richness of native birds were natural and heterogeneous landscapes, large cottonwood patches, close proximity to other cottonwood patches, and microhabitats with relatively open canopies. Landscape features were the most important and frequent predictors of distribution and abundance for most bird species, while macrohabitat and microhabitat were of secondary importance. Thus, landscape features should be a primary consideration for management of riparian habitats and selecting riparian reserve areas.

INTRODUCTION

Evidence of widespread population declines among many species of Neotropical migratory birds (Terborgh 1989, Hagan and Johnston 1992, Martin and Finch 1995, Rappole 1995) has intensified interest in avian conservation among scientists and land managers. The concern for Neotropical migrants (landbirds that breed mainly in temperate North America and winter primarily south of the United States-Mexico border [Finch and Stangel 1993]) first became heightened when Robbins et al. (1989a) reported that 75 percent of forest-dwelling migrants in eastern North America experienced population declines during the 1980s. Several human-caused factors (e.g, fragmentation and degradation of habitats) were indicated as operating to adversely affect populations of these species.

Although concern for these species originated from population monitoring of avifauna in the species-rich deciduous forest of the eastern United States, densities of breeding migrants are probably much higher in riparian habitats of western United States (Carothers and Johnson 1975, Ohmart and Anderson 1986, Knopf et al. 1988a, Finch 1991, Krueper 1993). These habitats comprise less than 1 percent of the landscape in the arid western United States, yet more species of breeding birds are found in this limited habitat compared to the more extensive surrounding uplands (e.g., Mosconi and Hutto 1982, Knopf et al. 1988a, Finch 1991, Saab and Groves 1992). Because of the presence of free water, riparian habitats in western North America have been greatly exploited and have suffered a century of degradation from livestock grazing, agriculture, water diversion, recreation, and other land-use activities (Thomas et al. 1979, Blakesley and Reese 1988, Sedgwick and Knopf 1989, Rood and Heinze-Milne 1989, Bock et al. 1993a, Malanson 1993, Ohmart 1994, Knight and Gutzwiller 1995, Saab et al. 1995). Western riparian areas are among the most threatened habitats on the continent because they are favored for these many land uses (cf. Terborgh 1989).

Because western riparian habitats are fragmented and limited in distribution, the total population numbers of migratory birds in these habitats probably are much smaller than those of migrants in woodlands of eastern North America (Terborgh 1989, Finch 1991). Consequently, western migratory landbirds may be particularly vulnerable to degradation of riparian woodlands. Thus, protection of existing healthy riparian woodlands and restoration of degraded or destroyed riparian systems should be a high priority for bird conservation in the western United States. Knowledge of bird responses to various land-use activities can provide critical insights into understanding and sustaining the integrity of riparian ecological systems.

Two major land-use activities altering the quality and quantity of riparian habitats at local, landscape, and regional scales are livestock grazing and recreational activities (e.g., Malanson 1993, Knight and Gutzwiller 1995, Saab et al. 1995). These activities may act alone to influence the persistence of landbirds, but it is also probable that they affect species by interacting together and with other forces (such as habitat fragmentation) in synergistic and cumulative ways.

Habitat fragmentation has been defined as "simply the disruption of continuity," which allows it to apply to any spatial scale (Faaborg et al. 1995). Fragmented habitats result in both quantitative and qualitative losses of habitat for species originally dependent on that habitat (Temple and Wilcox 1986). As a consequence, species abundance and diversity often decline and losses are greatest in smaller fragments (see Askins et al.1990).

Disturbances created by livestock grazing and recreational activities could exacerbate losses of plant and animals occupying small habitat fragments. Birds generally do not respond to the presence of grazing livestock but to the indirect impacts on vegetation as a result of grazing (Bock and Webb 1984). Grazing activities affect riparian habitats by altering, reducing, or removing vegetation, and by actually eliminating riparian areas through channel widening or lowering the water table (cf., Platts 1991, Mulchunas and Lauenroth 1993, Fleischner 1994).

Recreational activities not only affect birds through the indirect impacts of habitat modification but also directly by the presence of humans (Knight and Cole 1995a). Historically, the perception has been that outdoor recreation posed little environmental threat compared to extractive uses of natural resources such as timber harvest and livestock grazing (Flather and Cordell 1995). However, recreationists can degrade the land, water, and wildlife resources that support their activities by simplifying plant communities, increasing animal mortality, displacing and disturbing wildlife, and distributing refuse (Boyle and Samson 1985). Human-induced disturbance can have significant negative effects on breeding success by causing nest abandonment and increased predation (Hockin et al. 1992).

Several studies have evaluated the effects of cattle grazing on breeding bird communities in riparian habitats of western North America (e.g., Mosconi and Hutto 1982, Sedgwick and Knopf 1987, Knopf et al. 1988b, Schulz and Leininger 1991), whereas I am aware of only two studies that have examined the impacts of recreational activities on riparian avifauna (Aitchison 1977, Blakesley and Reese 1988). No prior study has investigated the influences of grazing and recreation in tandem, or in relation to forest patch dynamics, and these conditions often occur together in the western United States.

In this study, I evaluated bird and vegetation characteristics in cottonwood riparian forests, along the Snake River in Idaho, under three types of land use: (1) areas managed for cattle grazing [grazed]; (2) areas managed as campgrounds [recreation]; and (3) areas not managed for grazing or recreation but for riparian and wildlife habitat values [unmanaged]. Although landscape features surrounding forest patches are very important in explaining patterns of habitat use by breeding birds (e.g., Saab 1999, Freemark et al. 1995) and should be considered for certain types of analyses (particularly when making regional comparisons), my goal here was to examine, within a microhabitat (local) scale, the relative influence of grazing, recreation, and patch size on the distribution and abundance of breeding birds in riparian forest patches. Specific questions were: (1) How do bird species composition, richness, and abundance differ among lands managed for livestock grazing, recreation, and areas with little direct management? (2) Does riparian-forest patch size influence bird responses to these land-use practices? (3) Do bird species diversity, evenness, and turnover differ among land uses ? (4) Which individual bird species and nest guilds are favorably or negatively affected by the different land uses and patch sizes? and (5) Are structural patterns in vegetation layers among land uses and patch sizes similar to abundance patterns of birds occupying those vegetation layers?

STUDY AREA AND METHODS

Study Area Description

The study area encompassed the cottonwood riparian forests along 100 km of the South Fork of the Snake River (South Fork) in southeastern Idaho (Fig.1). Elevation ranges from 1700 m on the upstream end to 1460 m on the downstream end (confluence with the Henry's Fork of the Snake River). The climate is characterized by relatively low annual precipitation (550 mm), most of which comes in the form of snowfall during winter months.

The surrounding landscape is dominated by upland natural vegetation on the upstream end, and by agriculture on the downstream end (Saab 1992). Some cottonwood forests are naturally fragmented in the upper portions of the river, while others are fragmented as a result of agricultural development, especially downstream. Cottonwood fragments ranged in size from < 1 ha to > 200 ha.

The streamside vegetation is dominated by narrowleaf cottonwoods (*Populus angustifolia*) in the canopy, with the woody subcanopy/understory vegetation composed primarily of red-stemmed dogwood (*Cornus stolonifera*), and lesser amounts of thin-leaved alder (*Alnus incana*), water birch (*Betula occidentalis*), willows (*Salix* spp.), Rocky Mountain juniper (*Juniperus scopulorum*), silverberry (*Elaeagnus commutata*), chokecherry (*Prunus virginiana*), and hawthorne (*Crataegus* spp). The stream corridor is managed for irrigation, recreation, and livestock grazing.

Study Site Selection

Fifty-seven cottonwood forest patches (i.e., study sites) were located along a 100-km section of the South Fork. More than half (54%) of the cottonwood patches were created as a result of agricultural development; the remaining patches were created naturally by river channels. Selection criteria included land use activities, age and size classes of cottonwood patches, and isolation of cottonwood stands. Cottonwood forest patches were determined by delineating breaks in the forest canopy that were at least 100 m in width. All sampling areas were located in mature cottonwood stands. Each study site was managed for one of three types of land uses: (1) cattle grazing, (2) recreation campgrounds, and (3) unmanaged [areas not managed for grazing or recreation with little human use]. Land-use classes were selected on the basis of management activities. Actively managed sites were located in cottonwood patches where at least 75% of the area was used by livestock or public recreation. Unmanaged areas were not managed for livestock grazing or recreation and met the following criteria: (1) vegetation relatively undisturbed, (2) no obvious recent disturbance by humans, and (3) free from livestock grazing for at least three years. Cottonwood forests exclusive of livestock grazing and outside of designated campgrounds were managed for their riparian and wildlife habitat values.

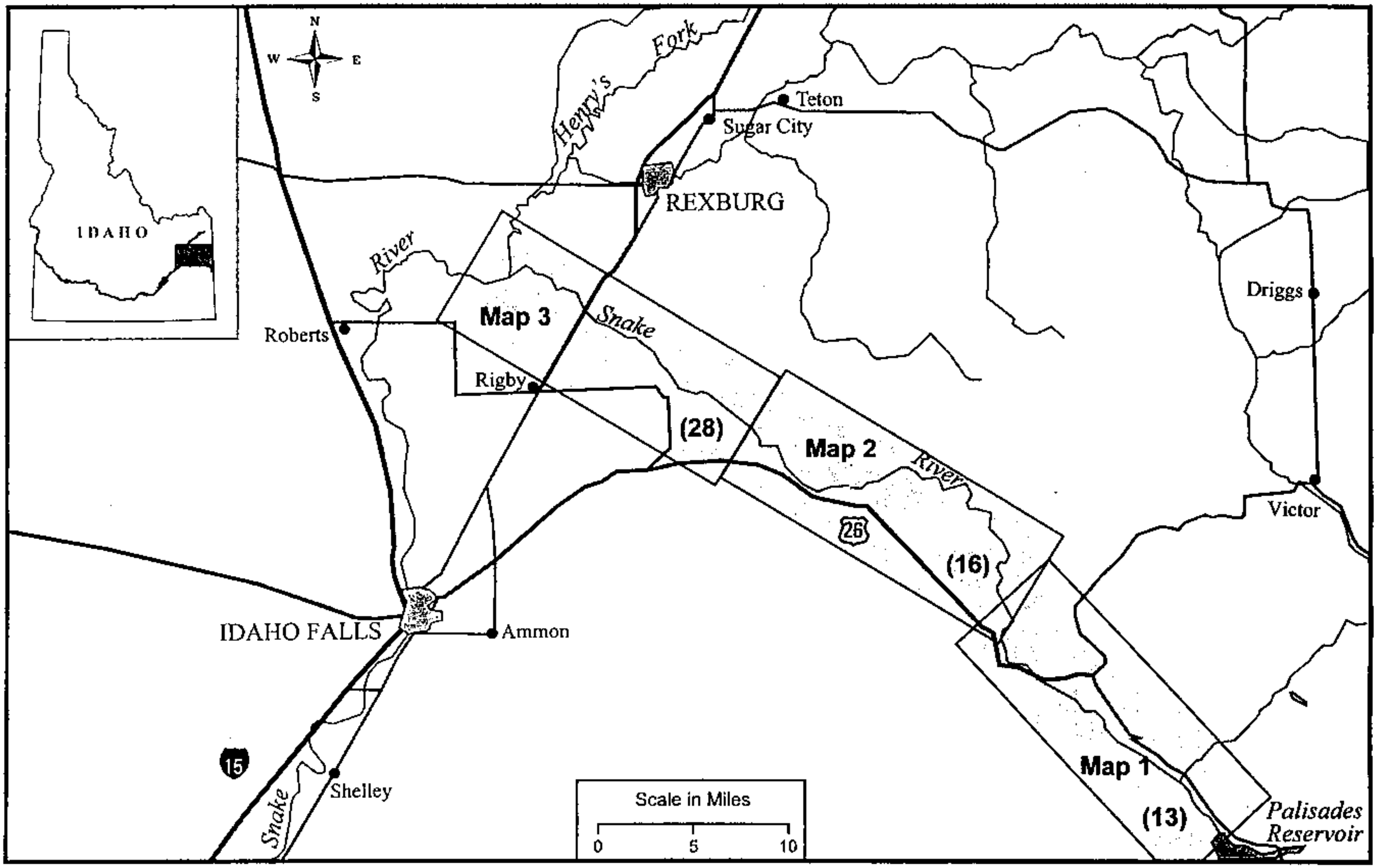

Figure 1. Location of the cottonwood riparian study area. Numbers in parentheses are the number of cottonwood patches in each map section that were sampled from 1991 - 1994.

Once I evaluated all cottonwood patches that met these criteria, sites were selected at random, except for recreation sites in small patches and all land uses in the large size class, where all available sites were selected for study. I avoided selecting sites managed for both grazing and recreation except for one large-patch recreation site that was grazed in autumn after the growing season and after migratory birds departed the study area. Riparian habitats grazed during autumn months (as compared to other seasons of the year) apparently have the least impact on breeding birds (Saab et al. 1995).

Table 1. Number of cottonwood patches and sampling stations within cottonwood patches used for surveying birds with point counts and measuring vegetation.

		Land Use		
Patch Size (ha)	Grazed	Recreation	Unmanaged	Totals
		No. Patches/*Stations*		
Small (<1-3)	13 / *17*	2 / *3*	13 / *21*	28 / *41*
Medium (>3-10)	3 / *8*	3 / *10*	9 / *25*	15 / *43*
Large (>10-200)	5 / *26*	4 / *20*	5 / *23*	14 / *69*
Totals	21 / *51*	9 / *33*	27 / *69*	57 / *153*

Each study site fit one of three patch-size classes (small, medium, and large; <1-3 ha, >3-10 ha, and >10-204 ha, respectively), resulting in nine sampling categories (Table 1). Size classes of cottonwood patches were based on the effects of patch size on breeding birds in riparian forests (Stauffer and Best 1980, Gutzwiller and Anderson 1987) and in forests fragmented by agriculture (Loman and von Schantz 1991). Within each size or land-use class (e.g., small grazed; see Table 1) there were at least 33 permanently marked stations, 150 m apart, from which birds were surveyed and vegetation was measured. This resulted in 153 sampling stations, with 84 stations in managed areas and 69 in unmanaged habitats.

Sites managed for livestock were variously grazed by cattle, and the timing and intensity of grazing were not controlled. For the previous five years and during the study, most (57%) sites were grazed throughout the growing season (May-September); some (29%) were grazed only in spring, summer, or fall; and a few (14%) were grazed on a rest-rotation basis (cf. Saab et al. 1995). Grazing intensity ("light," "moderate," or "heavy," based on the percentage of herbaceous vegetation consumed by livestock [Society of Range Management 1989]), varied among sites but was usually moderate (57%), often heavy (38%), and occasionally light (5%).

Campsites had been established for at least 10 years on all areas managed for recreation, and these sites were characterized by patches of trampled vegetation and/or bare soil. Most campgrounds were considered primitive because few or no facilities existed for recreationists. Recreational activities were primarily camping, day hiking, and fishing but at the most heavily used campgrounds (two of nine sites), recreation also included off-road driving. Most campgrounds were used only on weekends from late-spring throughout the summer. The number of recreationists varied among sites. On the two largest (> 10 ha) and most heavily used campgrounds, an average of 37.65 ($\pm$ 8.83 [1 SE]) recreationists were camped on a typical summer weekend (U.S.D.I. Bureau of Land Management, unpublished data).

Bird Surveys

Relative bird abundance was quantified using point-count surveys (Ralph et al. 1993) in 40-meter fixed-radius circular sampling stations (cf. Szaro and Jakle 1985) that were placed at least 20 m from a forest edge or in the center of the 57 cottonwood patches. The number of circular sampling stations varied depending on the area of the cottonwood patch, from one on sites of less than 4 ha to as many as 6 on a site more than 200 ha. On sites with more than one sampling station, station centers were separated by at least 150 meters. A total of 153 stations were sampled in this study. Observers surveyed birds for 10 minutes per visit at each station. Each station was visited twice during the 1991 breeding season, and three times during 1992-1994, from 15 May to 15 July. Bird counts were conducted by two people during 1991, and by three people during other years (1992-1994). Two of the same observers conducted point counts in three of four years. Each observer visited every station in each season in an attempt to minimize observer effects (Verner 1985).

Bird surveys were conducted between 06:00 and 11:00 and were confined to days with good weather (wind less than 20 mph and light or no precipitation). Stations were sampled at different times in the morning during the breeding season to reduce time-of-morning effects (Ralph et al. 1993). To reduce the bias of surveying at different periods of the breeding season, the first survey was conducted 15 May-4 June, the second survey from June 5-24, and the third survey from 25 June-15 July of each year. The total number of species, the number of individuals of each species, and the total number of individuals were recorded for each circular sampling station. Species and individuals recorded as flying over survey stations and species not breeding in the study area were excluded from analyses.

Thirty-four bird species known to nest in the study area were used in making estimates of total species richness and overall abundances for small landbirds (Appendix 1, Appendix 2). Thirty of those species were used for all analyses and were recorded in a minimum of 12 cottonwood patches and at least 14 sampling stations.

Habitat Measurements

Vegetation measurements were collected during 1991 and 1992 at the 153, 40-meter radius (0.5 ha) circular sampling stations used for bird surveys (see Table 1). Within each 0.5 ha sampling station, estimates of vegetation structure and composition were made in four 5-m-radius subcircles (0.008 ha). The initial subcircle was located in the center of the sampling station. The center of the next subcircle was located at a random compass direction and a fixed distance of 29 m from the station center. Each of the two remaining subcircles were positioned 120° from the first subcircle (cf. Ralph et al. 1993).

Stem densities of trees and shrubs were recorded by species and diameter size class at breast height. Woody vegetation was grouped into diameter size classes as follows: ≤2 cm, >2-5 cm, >5-8 cm within the 5-m-radius subcircle; and >8-23 cm, >23-38 cm, and >38 cm within a 11.3-m-radius circle extended from the 5-m-radius plot. Tree canopy was measured by using a densiometer at the center of each subcircle.

Ground cover was estimated on the 5-m radius subcircles by using an ocular tube (James and Shugart 1970). Ten readings were taken along transects using tape measures oriented parallel to the stream channel and the other perpendicular, such that they crossed at the center of the subcircle. Ground cover was estimated as percentages of either shrub, herbaceous, bare ground or litter based on frequency of occurrence at 2-m intervals along the tapes.

Data Analysis

Bird species richness, abundance, species diversity and eveness, and local species turnover were compared among grazed, recreation, and unmanaged cottonwood sites. Species diversity was calculated using the Shannon index of diversity [$H' = -\sum p_i \ln p_i$], where p_i is the proportion of individuals found in the *i*th species, and evenness [$E = H'/\ln S$], where S is total number of species (Magurran 1988). Local species turnover was determined by averaging the year to year change in species composition recorded over all sampling stations within a single cottonwood patch for three time periods: from 1991 to 1992, from 1992 to 1993, and from 1993 to 1994 (cf. Haila et al. 1993). Bird survey and vegetation data were combined by cottonwood patch to provide one sample per patch.

A two-way and three-way analysis of variance (ANOVA) or MANOVA for an unbalanced design (PROC GLM, SAS Institute, Inc. 1990) were the primary tests used for statistical comparisons of relative bird abundance and richness, bird species turnover rates, abundance of bird species grouped by nest layer (placement of nests as either ground, shrub, or canopy) and nest type (cavity or open), canopy coverage, woody-plant stem densities, and ground cover among land uses, patch sizes and in some cases among years. To test differences in species richness and abundance between various combinations of land use and patch size, I created an independent variable that had a level for each combination of the original independent variables (three land-use types and three patch-size classes) by concatenation (Cody and Smith 1991) and then performed a one-way ANOVA and multiple range tests for the main effects. Continuous variables used in (M)ANOVAs were tested for univariate normality, and arcsine (percentage data) or log (count data) transformed if necessary. Following transformation, all variables were normally distributed (Shapiro-Wilkes statistic, $P > 0.10$, SAS PROC UNIVARIATE). Type III Tests (SS [sums of squares] or SS&CP[cross products] matrices) were used for all (M)ANOVAs.

Wilk's Lambda was selected as the MANOVA test statistic. All multiple comparisons of main effects were computed with Tukey studentized range tests (SAS Institute, Inc. 1990). To determine if abundance of species grouped by nest layer was related to vegetation structure, vegetation layer measurements (ground cover, shrub densities, and canopy cover) were correlated with species' abundances in each nest layer (ground, shrub, canopy) using Spearman ranks.

Separate analyses on relative abundance of 30 individual species were completed for each species with numbers of detections large enough to fit regression models that included effects of land use and patch size (Table 2). Year was not included in the individual species models because few species had large enough sample sizes for valid results. Poisson regression (SAS/INSIGHT, SAS Institute, Inc. 1993) was used to test statistical differences in the mean number of detections (response variable) for each species per point count visit among land uses and patch sizes (the two main effects) and the interaction between the main effects. The interaction effect was not tested for seven species with small numbers of observations in each combination of land use/patch size, and a reduced model was fitted using only the main effects (Table 2). A Type III Wald Chi-square was the test statistic for the Poisson Regression. Poisson regression was appropriate for analyses of individual species because the response variable represented counts, including many zeros. For species in which the analysis showed a significant interaction effect between land use and patch size or a significant patch size effect, logistic regression was used to aid in examining the relationship between area (ha) and probability of the species' occurrence (Robbins et al. 1989b) within each type of land use. In the logistic regression analysis, an index of occurrence was calculated for each species for each cottonwood patch to serve as the dependent variable. In this calculation, the dependent variable assumes a value of 0 if the species was not detected on a point count visit to the sampling station and 1 every time it was detected on a single visit. Most sampling stations were visited 11 times over four breeding seasons, and the average number of occurrences per sampling station within a cottonwood patch was used to calculate a species' predicted probability of incidence within a cottonwood patch. Alpha-levels were set at 0.05 for all statistical tests.

Table 2. Mean number of detections per point count visit ($\pm$ 1 standard error) for each species preceded by the number of sites [ ] in which each species was recorded. The interaction effect between land use and patch size was not tested (NT) for species with few observations (see text for explanation). Numbers under each type of land use or patch size are the number of cottonwood patches sampled for that group. NS = not significant. For species with significant effects, different letters indicate that corresponding means are significantly different ($p \leq 0.05$) within land uses or within patch sizes. Common names of species' acronyms are listed in the Appendix 1.

Species	Land Use: Grazed (21)	Land Use: Recreation (9)	Land Use: Unmanaged (27)	*P*-value	Patch Size: Large (14)	Patch Size: Medium (15)	Patch Size: Small (28)	*P*-value	Interaction *P*-value
AMKE	[7] 0.08(0.03)	[4] 0.05(0.02)	[13] 0.06(0.02)	NS	[8] 0.07(0.02)	[7] 0.04(0.02)	[9] 0.07(0.02)	NS	NT
MODO	[20] 0.61(0.09)a	[8] 0.19(0.03)b	[22] 0.25(0.03)b	<0.001	[14] 0.37(0.04)a	[11] 0.22(0.04)b	[25] 0.44(0.07)a	0.034	0.002
RNSA	[14] 0.11(0.02)	[8] 0.16(0.03)	[21] 0.18(0.03)	NS	[14] 0.21(0.03)	[12] 0.15(0.03)	[17] 0.12(0.03)	NS	NS
DOWO	[15] 0.11(0.02)	[5] 0.04(0.01)	[18] 0.07(0.01)	NS	[15] 0.06(0.01)	[9] 0.09(0.02)	[14]0.08(0.02)	NS	NT
NOFL	[17] 0.23(0.03)	[8] 0.24(0.04)	[25] 0.18(0.03)	NS	[15] 0.26(0.04)	[12] 0.18(0.02)	[23] 0.19(0.03)	NS	NS
WWPE	[17] 0.23(0.03)	[4] 0.08(0.03)	[19] 0.20(0.03)	NS	[12] 0.20(0.03)	[9] 0.15(0.03)	[19] 0.22(0.03)	NS	NS
DUFL	[9] 0.06(0.02)a	[6] 0.04(0.01)b	[11] 0.03(0.01)b	0.002	[11] 0.05(0.01)a	[8] 0.08(0.02)a	[7] 0.02(0.01)b	0.004	NT
BBMA	[20] 0.53(0.09)a	[6] 0.30(0.09)ab	[20] 0.23(0.04)b	0.015	[15] 0.21(0.03)	[10] 0.32(0.09)	[21] 0.44(0.07)	NS	0.012
AMCR	[10] 0.11(0.03)	[5] 0.19(0.06)	[12] 0.10(0.02)	NS	[10] 0.17(0.05)	[8] 0.10(0.02)	[27] 0.09(0.02)	NS	NS
BCCH	[18] 0.20(0.03)a	[8] 0.24(0.05)ab	[25] 0.35(0.04)b	0.047	[15] 0.33(0.03)	[13]0.35(0.06)	[23] 0.22(0.04)	NS	NS
HOWR	[19] 1.15(0.09)a	[6] 0.17(0.04)b	[14] 0.46(0.06)b	<0.001	[13] 0.56(0.07)	[9] 0.46(0.08)	[17] 0.82(0.09)	NS	NS
GRCA	[9] 0.07(0.02)	[7] 0.15(0.04)	[21] 0.15(0.02)	NS	[12] 0.18(0.02)a	[11] 0.08(0.01)b	[14] 0.11(0.03)ab	0.025	NS
AMRO	[21] 1.34(0.09)	[8] 0.95(0.10)	[25] 1.19(0.08)	NS	[15] 1.08(0.09)	[13]1.11(0.10)	[26] 1.33(0.08)	NS	NS
VEER	[7] 0.11(0.03)a	[7] 0.25(0.05)a	[20] 0.48(0.05)b	0.048	[14] 0.38(0.05)	[8] 0.33(0.06)	[12] 0.28(0.05)	NS	<0.001
CEWA	[13] 0.20(0.04)	[7] 0.29(0.04)	[23] 0.31(0.05)	NS	[15] 0.24(0.03)	[13] 0.29(0.03)	[15] 0.27(0.05)	NS	NS
EUST	[19] 1.25(0.20)a	[4] 0.13(0.06)b	[21] 0.35(0.07)b	0.039	[13] 0.25(0.06)	[9] 0.18(0.05)	[22] 1.07(0.16)	NS	0.038

Table 2. Continued

Species	Land Use: Grazed (21)	Land Use: Recreation (9)	Land Use: Unmanaged (27)	P-value	Patch Size: Large (14)	Patch Size: Medium (15)	Patch Size: Small (28)	P-value	Interaction P-value
WAVI	[12] 0.35(0.05)a	[8] 1.00(0.09)b	[22] 0.49(0.05)a	<0.001	[15] 0.78(0.06)a	[12] 0.72(0.08)a	[15] 0.27(0.04)	<0.001	NS
YEWA	[20] 2.16(0.11)	[8] 2.13(0.17)	[27] 2.71(0.10)	0.049	[15] 2.67(0.12)a	[13] 2.37(0.14)ab	[27] 2.34(0.10)b	0.045	NS
YRWA	[11] 0.08(0.02)	[8] 0.20(0.04)	[13] 0.06(0.01)	NS	[14] 0.15(0.03)	[11] 0.13(0.02)	[7] 0.04(0.01)	NS	NS
MGWA	[6] 0.02(0.01)	[8] 0.15(0.04)	[12] 0.06(0.02)	NS	[10] 0.03(0.01)	[8] 0.10(0.03)	[8] 0.05(0.02)	NS	NS
YBCH	[3] 0.02(0.01)a	[2] 0.01(0.01)a	[7] 0.05(0.02)b	0.035	[6] 0.07(0.03)	[4] 0.04(0.02)	[2] 0.02(0.01)	NS	NT
WETA	[10] 0.05(0.01)	[7] 0.05(0.01)	[10] 0.03(0.01)	NS	[10] 0.06(0.02)	[8] 0.05(0.02)	[9] 0.03(0.01)	NS	NS
BHGR	[14] 0.13(0.02)	[8] 0.24(0.04)	[24] 0.23(0.03)	0.032	[14] 0.25(0.02)	[12] 0.20(0.03)	[20] 0.17(0.03)	0.049	NS
LZBU	[10] 0.06(0.02)a	[4] 0.07(0.02)ab	[20] 0.12(0.02)a	0.038	[11] 0.09(0.02)	[9] 0.09(0.02)	[14] 0.09(0.02)	NS	NT
NOOR	[20] 0.53(0.06)	[8] 0.25(0.05)	[26] 0.41(0.04)	NS	[15] 0.33(0.04)	[12] 0.35(0.05)	[27] 0.52(0.05)	NS	<0.001
BHCO	[19] 0.46(0.06)	[7] 0.17(0.04)	[24] 0.45(0.04)	NS	[15] 0.39(0.04)	[13] 0.36(0.05)	[22] 0.45(0.05)	NS	NS
CAFI	[9] 0.03(0.01)	[4] 0.03(0.02)	[6] 0.02(0.01)	NS	[10] 0.04(0.01)	[5] 0.02(0.01)	[4] 0.02(0.01)	NS	NT
AMGO	[21] 0.92(0.12)a	[8] 0.47(0.08)b	[27] 0.92(0.07)a	0.008	[15] 1.05(0.08)a	[13] 0.70(0.09)b	[28] 0.82(0.09)b	<0.001	0.003
FOSP	[3] 0.01(0.004)a	[4] 0.08(0.03)ab	[15] 0.15(0.03)b	<0.001	[10] 0.09(0.02)	[5] 0.07(0.02)	[7] 0.10(0.03)	NS	NT
SOSP	[13] 0.32(0.05)a	[9] 0.75(0.09)b	[24] 0.78(0.07)b	<0.001	[13] 0.45(0.05)a	[12] 0.63(0.07)ab	[21] 0.70(0.08)b	0.002	NS

RESULTS

Bird Detections

Observers recorded 16,850 individuals representing 34 bird species (Appendix 1) during 1,565 point-count visits distributed over 153 sampling stations placed among 57 cottonwood patches. Among these, 5,587 individuals were recorded during 511 visits in grazed areas; 3,171 individuals during 337 visits at recreational campgrounds; and 8,092 individuals in 717 visits at unmanaged sites. Most (82%) of the 34 species were Neotropical migratory landbirds (Appendix 1). Evaluating cumulative number of species over area revealed that detections of 32 species (94% of the species analyzed for this study) had accumulated by the time patch size reached 3 ha.

Overall species richness was similar among land-use types, with all 34 species recorded in grazed and unmanaged sites, and 32 species detected at recreational campgrounds (Eastern Kingbird and Yellow-billed Cuckoo were not recorded at campgrounds). Species diversity and evenness also were similar among land-use types (grazed H'=2.83, E=0.80; recreation H'=2.86, E=0.83; unmanaged H'=2.84, E=0.80). For cottonwood patches sampled in all four years (1991-1994), species turnover did not significantly differ among land uses [$\bar{x}$ ($\pm$1SE) for grazed [N=10]=1.53(0.04); recreation [N=5]=1.63(0.06); unmanaged [N=21]=1.61(0.03); df=2, F = 1.45, p = 0.25], where the numbers represent the average change in species composition from one year to the next recorded within a single cottonwood patch.

Land use had a very strong effect on mean number of species and individuals detected per point count visit (Table 3, Fig. 2). No overall size effect was found for number of species or individuals [$\bar{x}$($\pm$1SE) for number of species detected in large=4.12(0.14); medium=3.69(0.18); small=4.43(0.14) patches; and for number of individuals detected in large=11.15(0.32); medium=10.02(0.36); small=11.42(0.35) patches]. However, there was a significant interaction effect between land use and patch size. A significant year effect was found, while there was no interaction effect between year and the main effects of land use and patch size (Table 3). Tests of paired comparisons among land uses showed that mean number of species per point count visit was significantly different for all land uses, with species numbers lowest in recreation campgrounds (Fig. 2A). Mean number of individuals also was significantly reduced in recreation areas compared to grazed or unmanaged lands (Fig. 2B). Results of the multiple comparisons (an evaluation of the interaction between land use and patch size) revealed that mean number of species and individuals did not significantly differ between large-recreation patches and unmanaged areas, and large-recreation patches and the larger patch sizes of grazed lands (Fig. 3A, 3B).

Table 3a, 3b. Differences among effects of land use, patch size and year on (a) mean number of species per point count visit, and (b) mean number of individuals per point count visit.

a. Mean number of species per point count visit				
ANOVA for effects	SS	df	F	*p*
Main effects				
Land use	30.41	2	10.97	<0.001
Patch size	1.87	2	0.67	0.51
Year	14.20	3	3.42	0.02
Interactions				
Land use*Patch size	20.09	4	3.62	0.01
Land use*Year	2.16	6	0.26	0.95
Patch size*Year	4.77	6	0.57	0.75
Land use*Patch size*Year	7.24	10	0.52	0.87

b. Mean number of individuals per point count visit				
ANOVA for effects	SS	df	F	*p*
Main effects				
Land use	63.97	2	4.91	0.01
Patch size	26.32	2	2.02	0.14
Year	227.65	3	11.66	<0.001
Interactions				
Land use*Patch size	94.91	4	3.64	0.01
Land use*Year	29.18	6	0.75	0.61
Patch size*Year	19.54	6	0.50	0.81
Land use*Patch size*Year	15.14	10	0.23	0.99

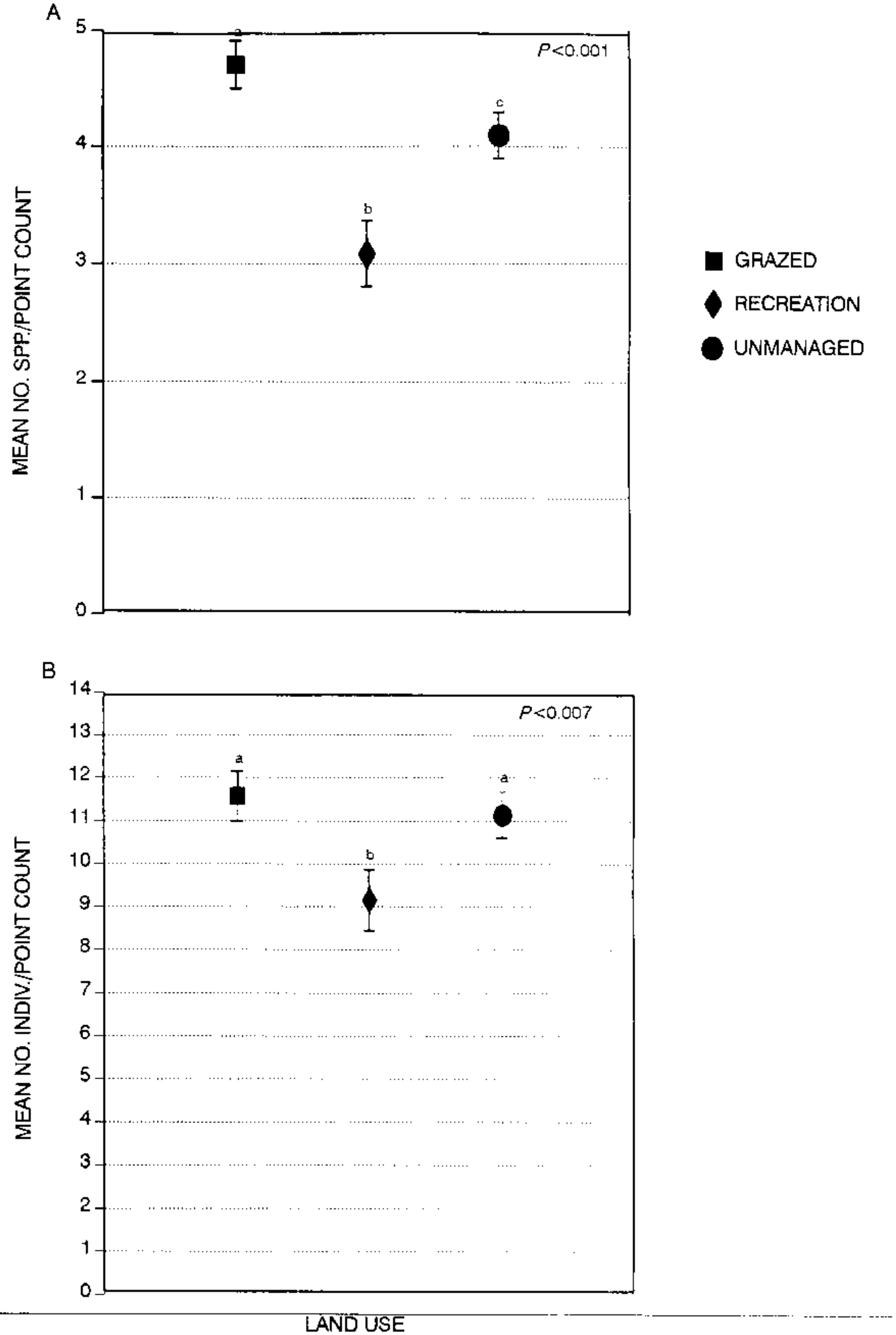

Fig. 2. Mean number of species (A) and individuals (B) detected per point count survey in each land-use type, averaged over all years and all patch sizes. Vertical lines represent $\pm$ SE. In each graph, different lower-case letters indicate that corresponding means are significantly different at $p \leq 0.05$.

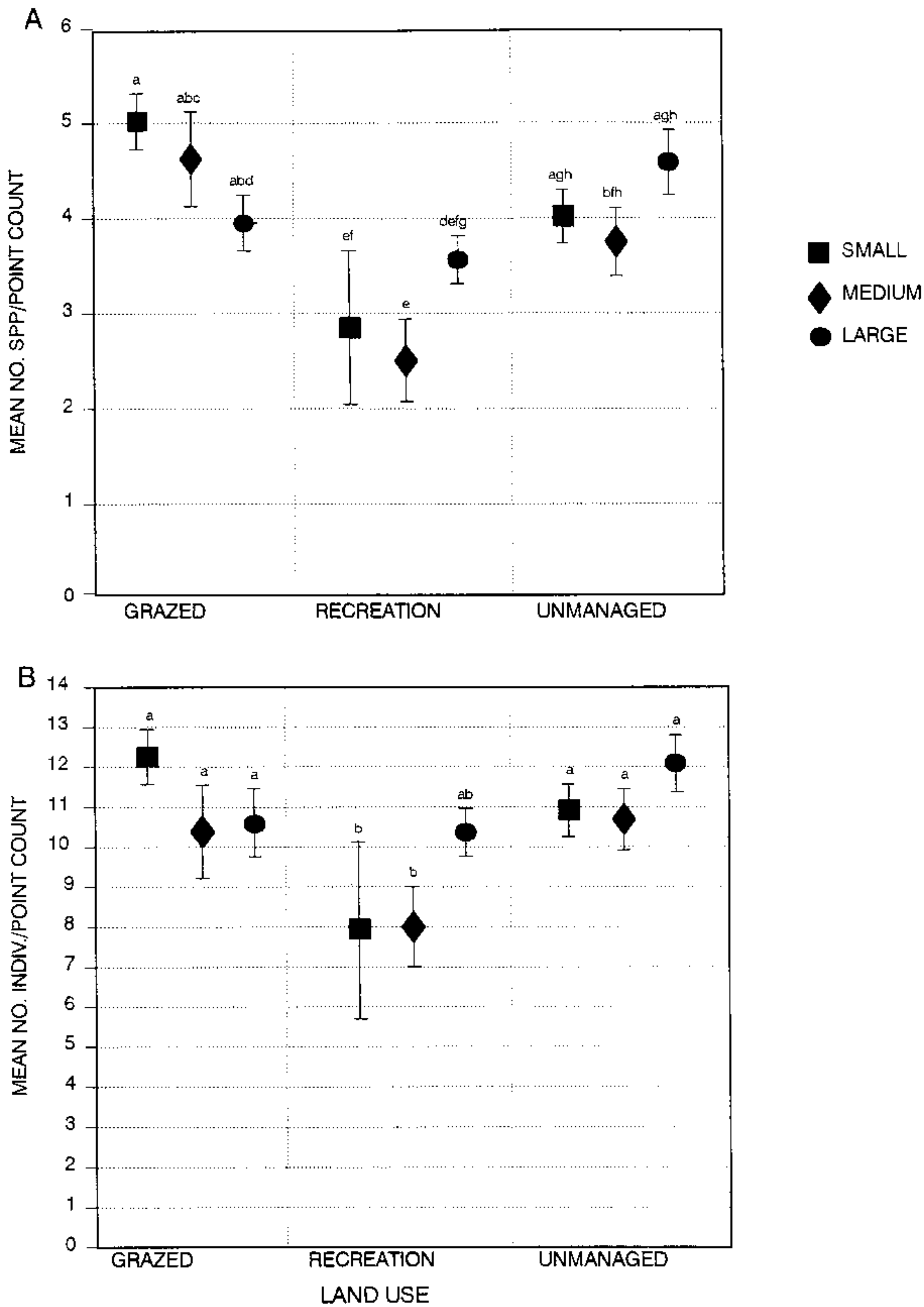

Fig. 3. Mean number of species (A) and individuals (B) detected per point count survey by land use and patch size, averaged over all years. Vertical lines represent ± 1 SE. In each graph, different lower case letters indicate that corresponding means are significantly different at $p \leq 0.05$.

Among years, 1991 generally had fewer species and individuals per point count visit compared to all other years (Fig. 4). Intensity of use by cattle or recreationists may have differed in some years but I did not quantify the change in use from one year to the next. During 1991 in unmanaged sites, however, species numbers and combined-individual abundances were significantly different from other years except for species richness in 1994. This suggests that a factor other than changes in land-use intensity may have been responsible for the reduced numbers of species and individuals during 1991 (e.g., local inclement weather).

Species were grouped by nest layer based on the placement of their nests (ground, shrub, or canopy; see Appendix 1) to test for differences in their relative abundance among land uses and patch sizes (Fig. 5A). Results suggested some differential land-use and patch-size effects among the three groups of nest layers (Table 4). Ground-nesting and canopy-nesting species were primarily responsible for the land-use effect, and ground nesters only for the patch-size effect (Table 4, Fig. 5). No effect was significant for shrub nesters, although their relative abundance was lowest in recreation areas (nonsignificant p=0.11). Using these results for prediction, ground nesters should respond negatively to grazing and camping activities, canopy nesters should respond positively to grazing (at least over the short-term), and shrub nesters should tend to respond negatively to recreational and grazing activities.

Species were also grouped by nest type as either cavity or open-cup nesters (Appendix 1) to test for differences in their relative abundance among land uses. Open-cup nesters included a group of 26 species that varied greatly in their life histories and habitat requirements (e.g., American Crow and Yellow Warbler); however, the cavity-nesting group consisted of only eight species that did not vary as widely in life history, habitat use, or taxonomy (included four woodpecker species in the family Picidae). Results indicated a significant overall land-use effect on cavity and open-cup nesting species (Wilk's lambda = 0.76; $F(94,106) = 3.99$; $p = 0.005$). By examining the univariate ANOVAs and mean values for relative abundances of cavity and open-cup nesters, the differential response to land uses appeared largely due to cavity-nesting species, whose relative abundances were highest in grazed areas and lowest in recreation campgrounds (Table 5).

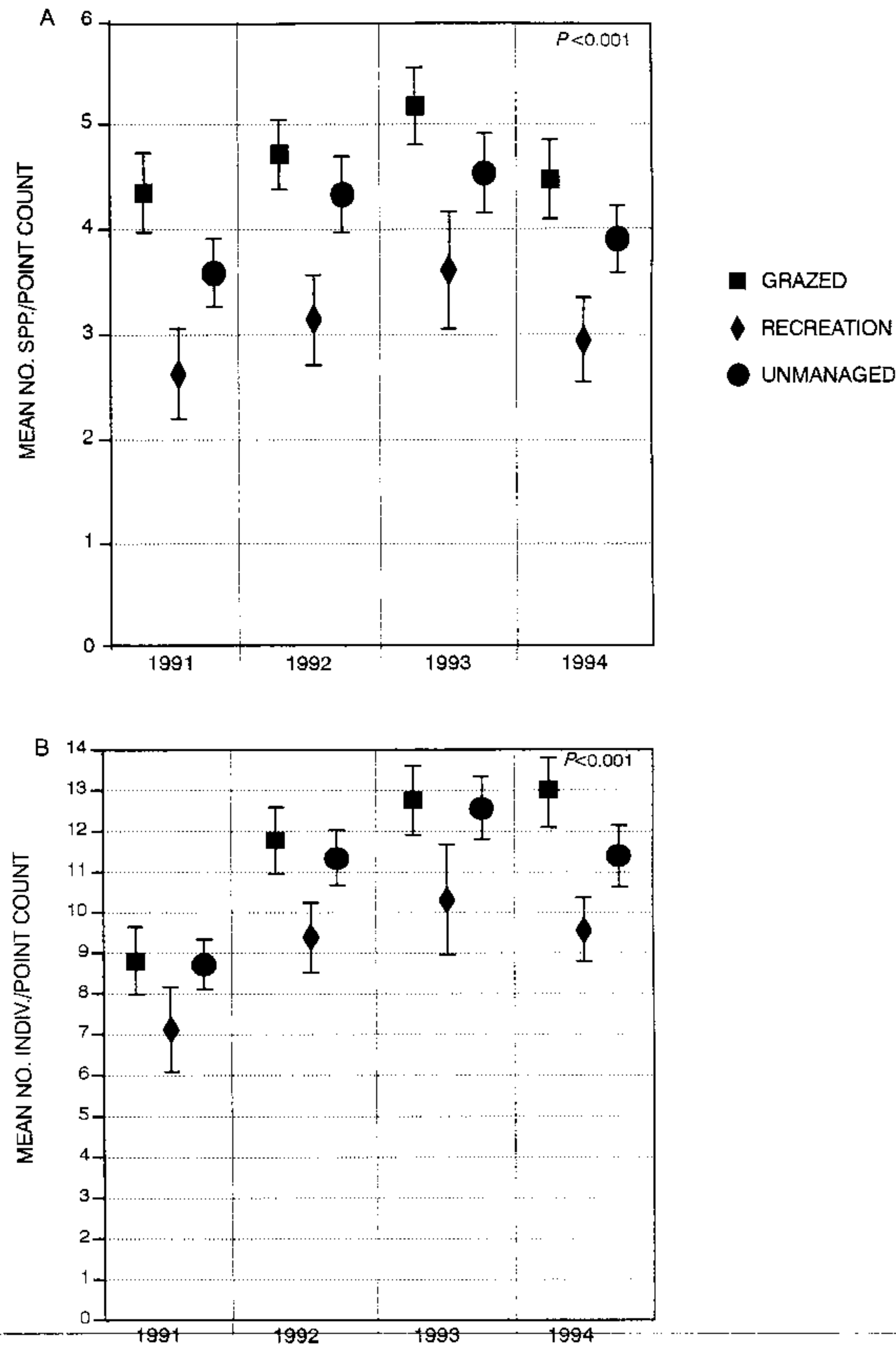

Fig. 4. Mean number of species (A) and individuals (B) detected per point count survey by year and land use, averaged over all patch sizes. Vertical lines represent $\pm$ 1 SE.

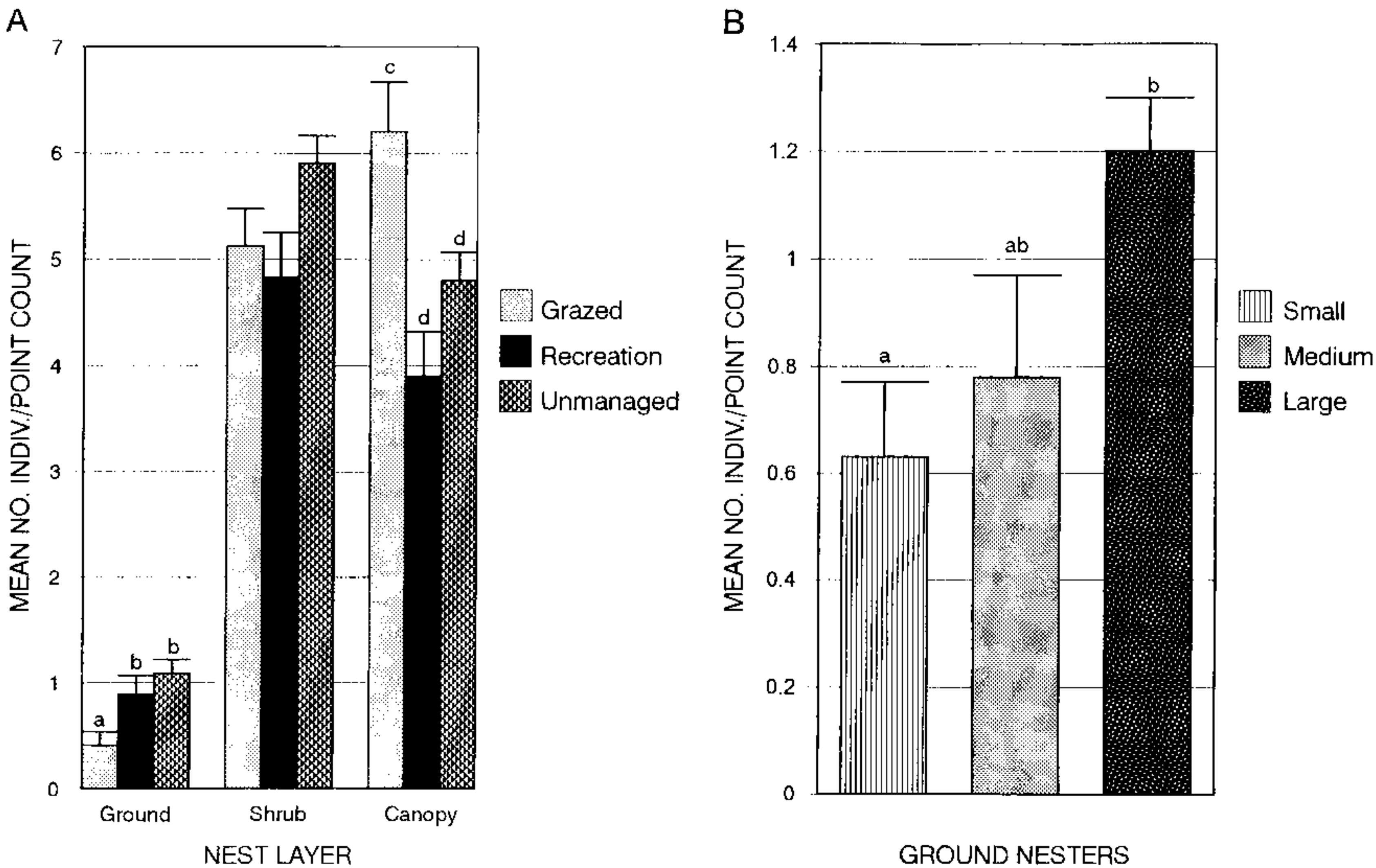

Fig. 5. Mean number of individuals by nest layers within each land-use type (A) and mean number of individuals for ground-nesting species within patch sizes (B). Vertical lines represent 1 SE. Within each nest layer, different lower-case letters indicate that corresponding means are significantly different at $p \leq 0.05$.

Table 4. Effects of land use and patch size on the abundance of species grouped by nest layer, based on the placement of their nests on either the ground, in shrubs, or in the canopy (see Appendix 1). ANOVA results are presented to aid in interpreting dependent variables from suggestive MANOVAs.

MANOVA effect	Wilk's Lambda	F	Num df	Den df	*p*
Land use	0.64	3.89	6	92	0.002
Patch size	0.65	3.69	6	92	0.003
Land use*Patch size	0.65	1.80	12	122	0.06

ANOVAs for effects	SS	df	F	*p*
Ground Nesters				
Land use	1.24	2	5.51	0.007
Patch size	1.27	2	5.67	0.006
Land use*Patch size	0.94	4	2.11	0.10
Error	5.38	48		
Shrub Nesters				
Land use	0.33	2	2.27	0.11
Patch size	0.02	2	0.15	0.86
Land use*Patch size	0.10	4	0.03	0.85
Error	3.52	48		
Canopy Nesters				
Land use	0.93	2	6.85	0.002
Patch size	0.35	2	2.59	0.09
Land use*Patch size	0.59	4	2.14	0.09
Error	3.28	48		

Over half (17 or 53%) of the 30 bird species analyzed suggested some differential effects of land use and patch size, and/or the interaction between the two main effects (Table 2). Compared to unmanaged sites with little or no use by cattle or recreationists, relative abundance per point count visit for eight species decreased in grazed lands (Black-capped Chickadee[BCCH], Veery[VEER], Yellow Warbler[YEWA], Yellow-breasted Chat [YBCH], Black-headed Grosbeak[BHGR], Lazuli Bunting[LZBU], Fox Sparrow [FOSP], and Song Sparrow[SOSP]). Abundance for eight species decreased in recreation campgrounds (Mourning Dove[MODO], Black-capped Chickadee[BCCH], House Wren[HOWR], Veery[VEER], Yellow Warbler[YEWA], Lazuli Bunting [LZBU], American Goldfinch[AMGO], and Fox Sparrow[FOSP]). In contrast, some species increased in grazed areas (Mourning Dove[MODO], Dusky Flycatcher [DUFL], Black-billed Magpie[BBMA], House Wren[HOWR], and European Starling [EUST]), and in recreation campgrounds (Warbling Vireo[WAVI]) compared to unmanaged areas. For most species showing significant patch-size effects, relative numbers were highest in large patches (Gray Catbird[GRCA], Warbling Vireo [WAVI], Yellow Warbler[YEWA], Black-headed Grosbeak[BHGR], and American Goldfinch [AMGO]).

Table 5a, 5b. Effects of land use activities on the relative abundance of species grouped by nest type (cavity or open-cup nesters). For descriptive statistics in each nest type category, different letters indicate that corresponding means are significantly different ($p \leq 0.05$).

a. Descriptive Statistics

Nest Type	Grazed Mean(±SE)	Recreation Mean(±SE)	Unmanaged Mean(±SE)
Cavity	3.28 (±0.42)a	1.70 (±0.29)b	2.26 (±0.17)b
Open	7.95 (±0.44)	7.76 (±0.70)	9.20 (±0.39)

b. Mean number of detections per point count visit

ANOVAs for effects	SS	df	F	*p*
Cavity Nesters				
Land use	19.94	2	5.39	0.007
Error	99.91	54		
Open-cup Nesters				
Land use	24.68	2	2.98	0.06
Error	223.74	54		

Results of the logistic regression suggested differential area effects on 10 species, depending on land use activities (Table 6). European Starling and Song Sparrow were the only species with probability of occurrences that were consistently and significantly highest in small patches within all land uses. Four species (Warbling Vireo [WAVI], Veery [VEER], Black-headed Grosbeak [BHGR], and Gray Catbird [GRCA]) that had no overall area relationship when land uses were combined, showed an area effect (i.e., increased probability of occurrence with increased patch size) in recreation sites and sometimes in grazed lands. Five species (American Goldfinch [AMGO], Yellow Warbler [YEWA], Veery [VEER], Black-headed Grosbeak [BHGR], and Gray Catbird [GRCA]) were unaffected by patch size in unmanaged areas, but showed significant area effects in grazed and/or recreation sites. Probability of occurrences for Northern Orioles and Black-billed Magpies were significantly highest in small patches of grazed areas, yet in unmanaged lands their probabilities were highest in large patches. Maybe results for these two species are due to chance effects, given the large number of statistical comparisons.

Habitat

Vegetation was measured at three layers: ground, shrub/subcanopy, and canopy. An overall land-use effect was found for three ground cover variables of bare, shrub, and herb (Wilk's Lambda = 0.63; $F(6,92) = 4.01$; $p = 0.001$) (Fig. 6), an overall patch size effect (Wilk's Lambda = 0.63; $F(6,94) = 4.02$; $p = 0.003$), but no interaction effect between land use and patch size. Percentage of bare ground and herbaceous cover increased with grazing and recreational uses, whereas shrub cover decreased with these same land-use activities (Fig. 6). Percentage bare ground [$\bar{x}(\pm$ 1SE) for large=9.02(2.02); medium=6.66(2.37); small=10.77(2.53) patches] and herbaceous cover [$\bar{x}(\pm$ 1SE) for large=19.37(1.80); medium=18.37(2.62); small=30.09(3.70)] also increased in small patches compared with medium and large patches. Percentage of ground covered by logs and litter did not differ significantly among land uses (Fig. 6) or patch sizes.

Densities of woody stems of various sizes were estimated within the shrub/subcanopy layer (Fig. 7). MANOVA results indicated an overall land-use effect on their densities (Wilk's Lambda = 0.60; $F(12,86) = 2.11$; $p = 0.02$), primarily due to reductions of smaller diameter stems in areas managed for grazing and recreation (Fig. 7a). There was no patch size effect ($p = 0.21$) or land use/patch size interaction ($p = 0.37$) on overall stem densities.

Stem densities were also recorded by plant species (Table 7). Overall stem densities of the most abundant woody plant species were apparently affected by land use activities and patch sizes (Table 7a). Changes in stem densities associated with land use effects were primarily due to reductions of alder, birch, and dogwood in grazed lands compared to unmanaged areas (Table 7b). Silverberry and Western Clematis were the only species with significantly different densities within patch size classes. Densities of both species were higher in large forest patches.

Table 6. Effect of patch size and land use on species detection. Values are probabilities of detecting species in cottonwood forest patches of various sizes within three land uses. *P*-values associated with each land use and the chi-square statistic for testing the relationship between species' frequency of occurrence and forest patch size of all land uses were estimated by logistic regression analysis. Species are ordered from those with the strongest positive relationship with patch size to those with the most negative relationship. Patch sizes are S=small (<1-3 ha); M= medium (>3-10ha); and L=large (>10-204 ha). *P*-value for chi-square test statistic: *$p<0.05$, **$p<0.01$, ***$p<0.001$; NS=nonsignificant.

	Probability of Detecting Species by Cottonwood Patch Size												
	Grazed				Recreation				Unmanaged				All Land Uses
Species	S	M	L	*p*	S	M	L	*p*	S	M	L	*p*	χ^2
AMGO	0.46	0.46	0.57	<0.001	0.26	0.26	0.34	0.07	0.54	0.54	0.56	0.64	11.57*
MODO	0.30	0.30	0.31	0.83	0.09	0.10	0.16	0.03	0.15	0.20	0.40	<0.001	10.05**
YEWA	0.92	0.92	0.94	0.08	0.75	0.83	0.96	<0.001	0.94	0.94	0.96	0.15	6.47*
DUFL	0.09	0.09	0.08	0.46	0.02	0.02	0.05	0.01	0.02	0.02	0.06	<0.001	3.67*
WAVI	0.37	0.38	0.39	0.52	0.59	0.61	0.70	0.01	0.35	0.38	0.50	<0.001	3.10(NS)
VEER	0.12	0.12	0.20	<0.001	0.15	0.17	0.26	0.004	0.37	0.37	0.34	0.51	0.02(NS)
BHGR	0.14	0.14	0.14	0.89	0.11	0.12	0.23	<0.001	0.20	0.20	0.20	0.94	0.13(NS)
GRCA	0.10	0.10	0.07	0.14	0.06	0.08	0.18	<0.001	0.12	0.12	0.15	0.20	0.32(NS)
NOOR	0.34	0.33	0.23	0.002	0.13	0.14	0.21	0.01	0.28	0.30	0.41	<0.001	4.29*
BBMA	0.26	0.23	0.09	<0.001	0.21	0.19	0.11	0.01	0.10	0.11	0.16	0.03	13.24***
EUST	0.32	0.32	0.14	<0.001	0.09	0.08	0.04	0.05	0.19	0.17	0.12	0.04	17.74***
SOSP	0.38	0.37	0.14	<0.001	0.63	0.60	0.34	<0.001	0.55	0.53	0.46	0.02	73.12***

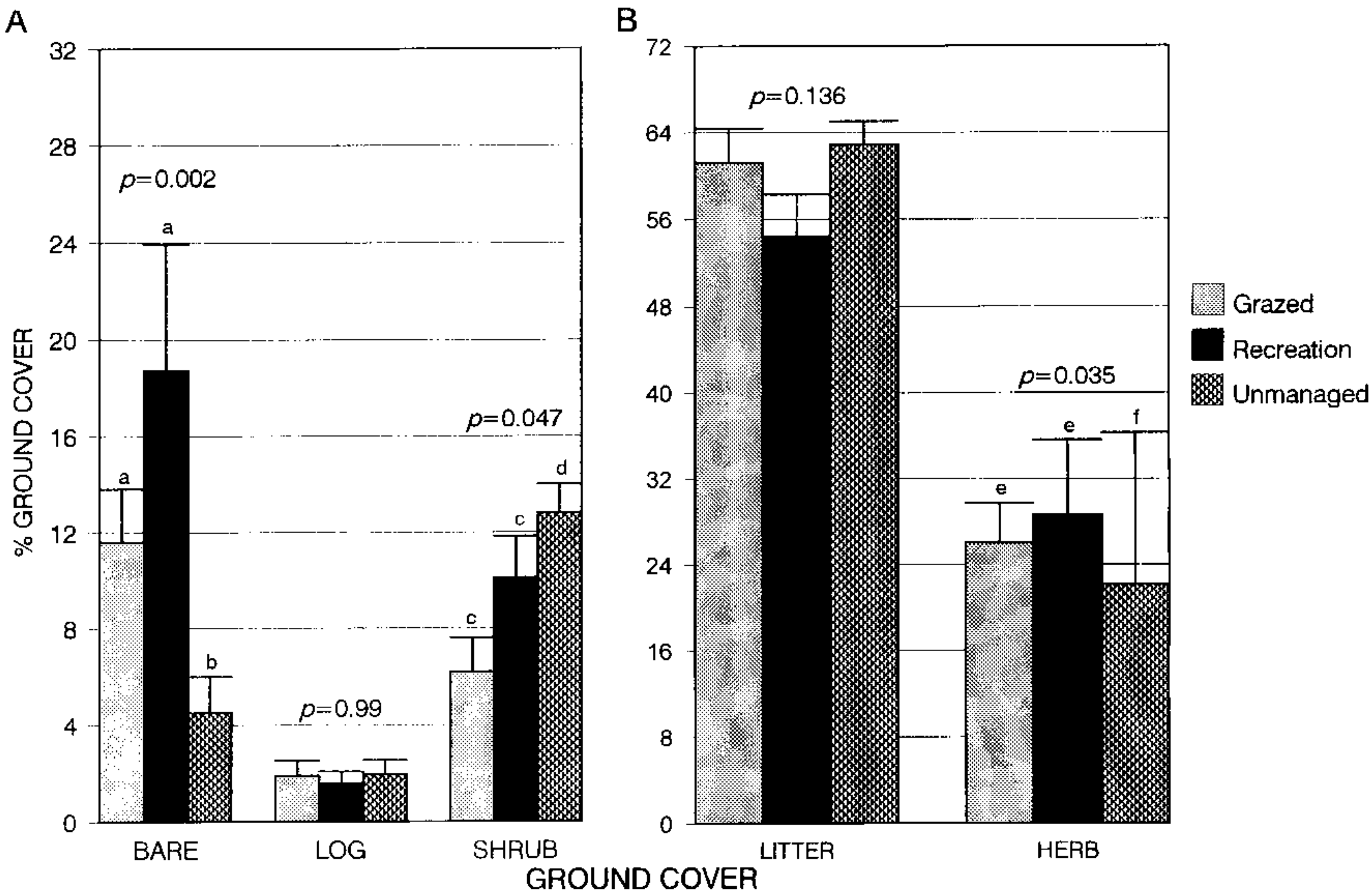

Fig. 6. Mean percentage of five ground cover categories within each land-use type. Vertical lines represent 1 SE. Within each ground cover category, different lower-case letters indicate that corresponding means are significantly different at $p \leq 0.05$.

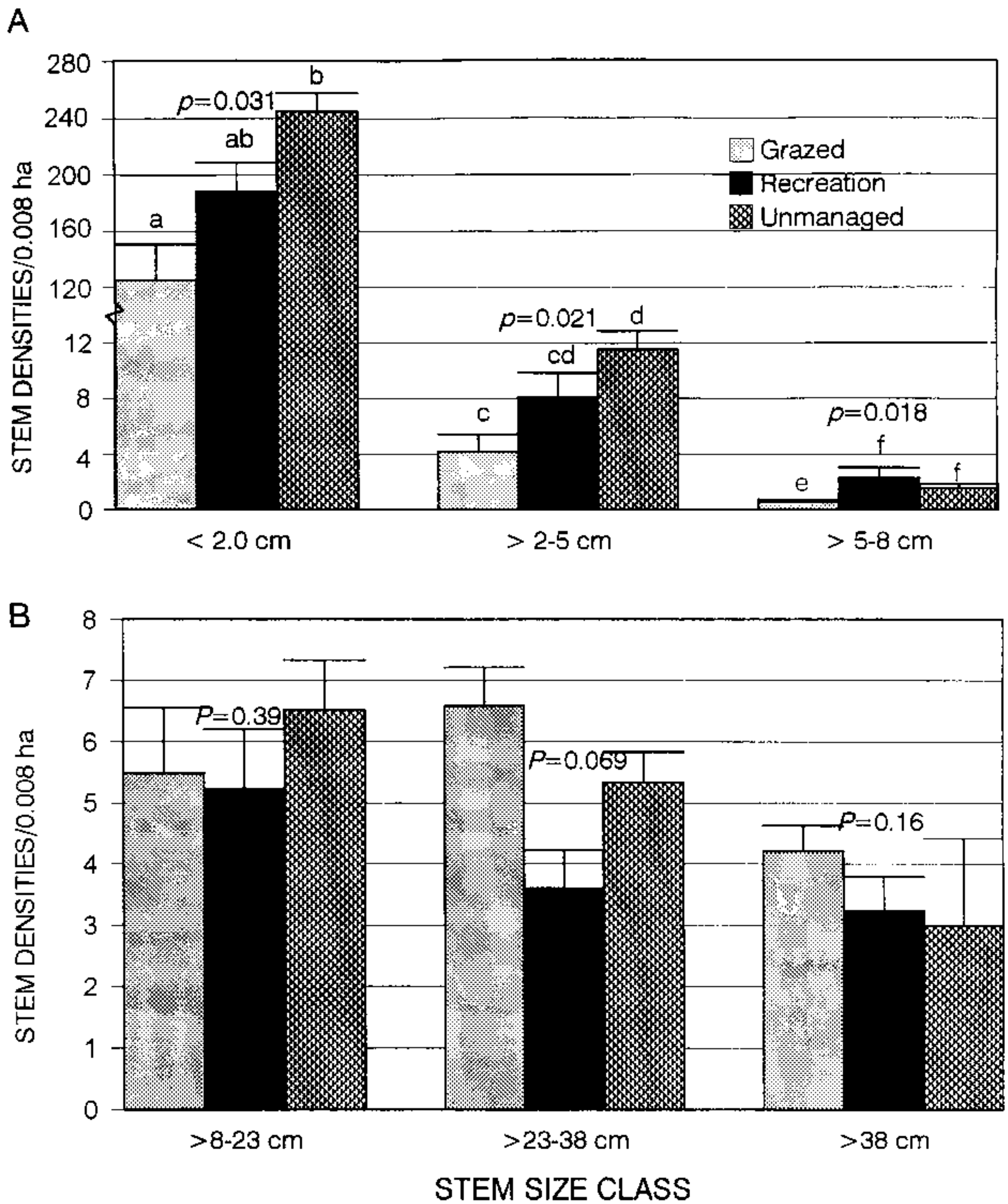

Fig. 7. Average stem densities for small stems (A) per 5-m radius (0.008 ha) sampling station, and large stems (B) per 11.3-m radius (0.04 ha) sampling station within each land-use type. Vertical lines represent 1 SE. Within each stem size class, different lower-case letters indicate that corresponding means are significantly different at $p \leq 0.05$.

Table 7a, 7b. Summary statistics, ANOVA, and MANOVA results for overall stem densities reported for the most abundant woody plant species by land use and patch size.

a. Mean (± 1 SE) and ANOVA results for stem densities of all sizes combined, averaged per 0.04 ha sampling station.

Species	Land Use				Patch Size			
	Grazed	Recreation	Unmanaged	*P*-value	Small	Medium	Large	*P*-value
Thin-leaved Alder *Alnus incana*	0.29(0.29)a	2.77(1.65)ab	3.81(1.32)b	0.03	1.48(0.58)	5.23(2.64)	1.53(0.68)	NS
Water Birch *Betula occidentalis*	2.95(1.75)a	45.62(12.20)b	12.29(3.74)a	<0.001	8.33(3.49)	17.62(9.67)	22.24(5.72)	NS
Western Clematis *Clematis ligusticifolia*	41.39(11.63)	72.84(24.93)	55.24(12.13)	NS	32.90(11.05)a	79.24(19.79) ab	68.81(11.25)b	0.004
Red-stemmed Dogwood *Cornus stolonifera*	42.47(10.89)a	79.55(19.37)ab	110.35(11.12)b	0.03	66.95(12.95)	98.81(14.88)	90.74(13.11)	NS
Douglas Hawthorne *Crataegus douglasii*	1.99(0.94)	1.29(0.45)	2.26(1.44)	NS	1.55(1.34)	2.11(1.14)	2.80(0.89)	NS
Silverberry *Elaeagnus commutata*	11.11(5.41)	11.36(6.87)	11.99(2.25)	NS	6.16(2.09)a	5.84(2.42)a	28.18(6.70)	<0.001
Rocky Mountain Juniper *Juniperus scopulorum*	1.95(0.66)	2.80(0.99)	2.40(1.23)	NS	1.32(0.44)	1.40(0.53)	4.97(2.16)	NS
Narrowleaf Cottonwood *Populus angustifolia*	23.56(4.09)	19.72(3.65)	16.43(1.27)	NS	20.52(3.18)	19.21(2.29)	18.07(1.73)	NS
Willow species *Salix* spp.	10.79(2.54)	23.09(8.53)	21.56(3.72)	NS	14.29(3.21)	24.07(5.04)	19.27(5.39)	NS

b. MANOVA on average stem densities

MANOVA	Wilk's Lambda	F	Num df	Den df	*p*
Land use	0.44	2.27	18	80	0.01
Patch Size	0.38	2.81	18	80	0.01
Land use*Patch Size	0.44	1.04	36	151.64	0.42

Minimal differences in percent canopy coverage were found among land uses (Fig. 8a), but overall canopy increased significantly with decreasing patch size (Fig. 8b). Of the eight plant species whose canopy reached the overstory [listed in decreasing order of canopy coverage: narrowleaf cottonwood, red-stemmed dogwood, water birch, willow spp., thin-leaved alder, chokecherry, Rocky Mountain juniper, and silverberry] no individual plant species had a significant canopy increase in small patches or significant changes among land uses.

Vegetation characteristics of the ground and shrub layers were significantly correlated with abundance of ground and shrub nesting birds, respectively (Table 8). Significant negative correlations were found between abundance of ground-nesting species and percentage of bare ground, and between ground nesters and percent canopy coverage. Significant positive correlations were found between shrub nesters, shrub cover, and shrub densities. Ground nesters also were positively correlated with shrub densities. No significant correlation was found between percentage of canopy cover and abundance of canopy-nesting birds. Ground- and shrub-nesting species showed significant positive correlations, whereas a significant negative correlation was found between ground and canopy nesters.

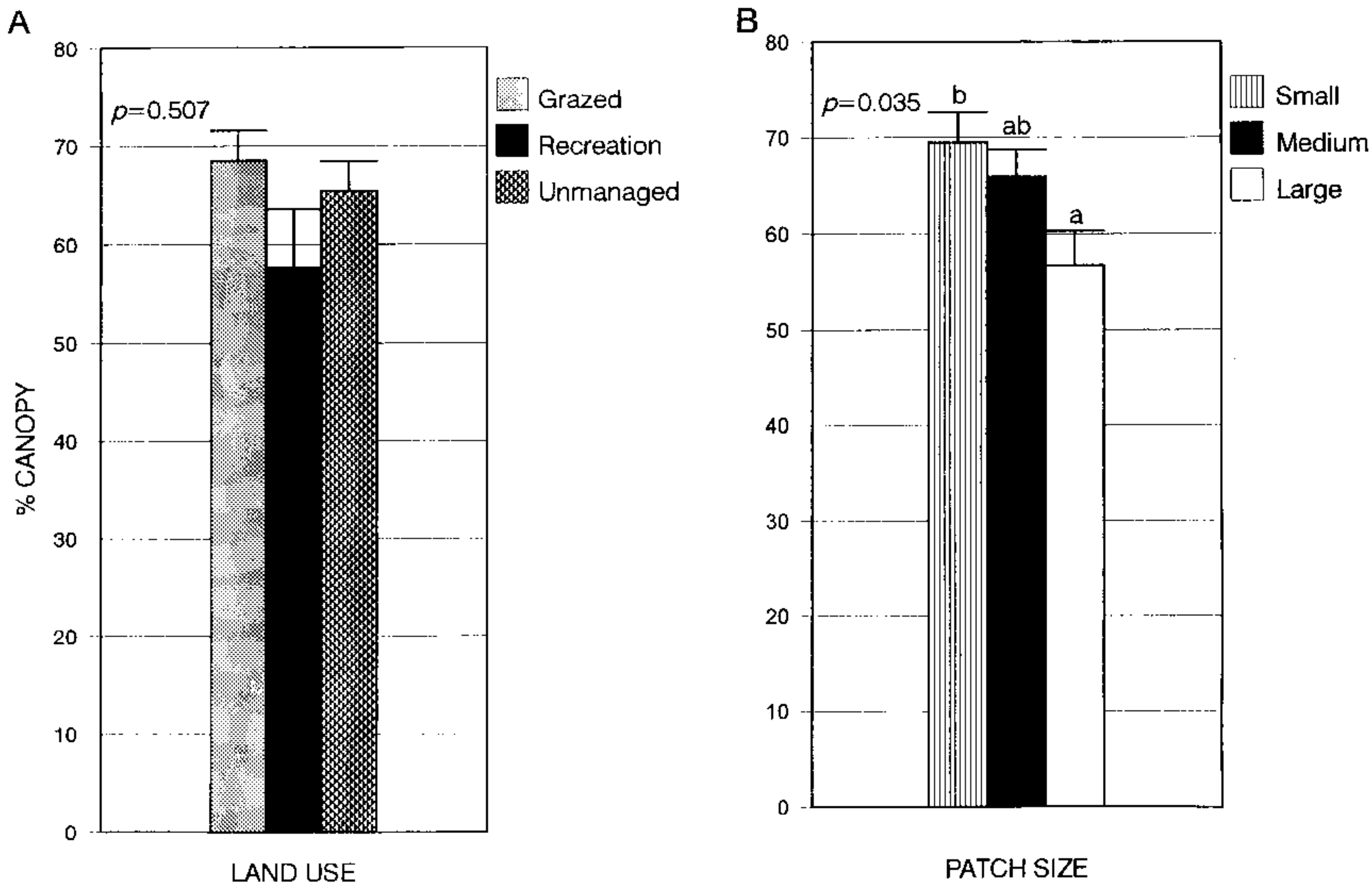

Fig. 8. Mean percentage of canopy cover in each land-use type (A) and patch-size class (B). Vertical lines represent 1 SE. Within each graph, different lower-case letters indicate that corresponding means are significantly different at $p \leq 0.05$.

Table 8. Correlation analysis of relative abundances of species grouped by nest layer (ground, shrub, canopy) and vegetation structural characteristics by layer (ground cover, shrub cover and densities, canopy cover) per sampling station. Spearman correlation coefficients are listed with statistical significance indicated by * $p \leq 0.05$, ** $p < 0.01$, *** $p < 0.001$.

	Ground Nesters	Shrub Nesters	Canopy Nesters	%Bare Ground	%Shrub Cover	Shrub Stem Densities ($\leq$ 2.5 cm)	Shrub Stem Densities (> 2.5 - 5 cm)
Shrub Nesters	0.33 *						
Canopy Nesters	-0.27 *	0.07					
% Bare Ground	-0.30 *	-0.36 **	0.05				
% Shrub Cover	0.24	0.40 **	-0.08	-0.33 *			
Shrub Stem Densities ($\leq$ 2.5 cm diameter)	0.34 **	0.25*	-0.14	-0.51 *	0.64 ***		
Shrub Stem Densities (>2.5 - 5 cm diameter)	0.54 ***	0.48 ***	-0.37 **	-0.44 ***	0.48 ***	0.43 ***	
% Canopy	-0.28 *	-0.11	0.21	0.09	0.03	-0.26	-0.12

DISCUSSION

Long-term effects of grazing, recreational activities, and habitat fragmentation on populations and communities can include changes in abundance, distribution, and demographics of populations, or changes in interactions and species composition of bird communities (cf. Knight and Cole 1995a). With disturbances by livestock and recreationists I predicted changes in overall species richness, diversity, evenness, and turnover, but these factors remained fairly constant in all areas along the South Fork of the Snake River. On average, species numbers and relative abundance appeared to be most reduced by recreational activities except in large patches managed for recreation. Although species composition differed between grazed and unmanaged sites, few statistical differences were found in overall mean number of species or mean number of individuals per survey visit.

Distribution and relative abundance of individual species, species grouped by nest layer, and species grouped by nest type, however, varied significantly among land-use activities and patch sizes. Abundances of birds nesting in canopy, shrub, and ground layers were positively correlated with abundance of these structural components of the vegetation. Timing and intensity of livestock grazing (Bock et al. 1993a, Milchunas and Lauenroth 1993, Saab et al. 1995) and recreational uses (van der Zande et al.1984, Knight and Cole 1995b) can have differential effects on plant and animal communities. Because these factors were not controlled, and varied within my study sites, some effects of grazing and recreational activities on bird community characteristics may have been masked.

Vegetation characteristics were most similar between recreation and unmanaged sites, yet overall bird abundance was lowest in recreation campgrounds. This suggests that additional factors, most likely the presence and activities of humans, may have had a prevailing influence on patterns of bird microhabitat use within recreational areas. If recreational disturbance is a primary reason for reduced abundances, then we need to know the relationships between disturbance, long-term persistence of avian populations, reproduction, and survival.

Wildlife responses to recreational disturbance are influenced by many factors, including the type of activity (e.g., motorized vs. nonmotorized), recreationist's behavior (e.g., slow vs. rapid movement), predictability (e.g, consistent vs. erratic), and timing (e.g., breeding season vs. nonbreeding) (Hockin et al. 1992, Knight and Cole 1995b). Little information is available about the consequences of these different influences on birds (Knight and Cole 1995a). However, greater impacts have generally been reported for recreation that is motorized (e.g., Titus and VanDruff 1981), rapid moving (e.g., Burger 1981), unpredictable (e.g., Klein 1993), and prevalent during the breeding season (e.g., Hockin et al. 1992). Most of the recreational activities within my study sites were nonmotorized and slow moving (camping, hiking, and fishing), and consistent only on weekends throughout the breeding season.

Other studies have evaluated recreational effects on bird community structure by comparing the avifaunal use of undeveloped areas and sites variously developed as campgrounds (Aitchison 1977, Foin et al. 1977, Robertson and Flood 1980, Clark et al. 1984, Blakesley and Reese 1988). Most of these studies generally found a higher species diversity in disturbed habitats, which was mainly due to additional, common opportunistic species moving into recreation areas, while some other species were reduced or eliminated by recreational development. One study reported lower bird species diversity and evenness in recreational developments but greater overall abundance than undeveloped sites, while species richness was similar between the two areas (Robertson and Flood 1980). Studies of recreation effects on bird populations in the Netherlands found that most species had significantly lower densities in developed parks with heavy recreational use (van der Zande and Vos 1984, van der Zande et al. 1984).

Along the South Fork of the Snake River, overall abundance was significantly reduced in recreation areas, while species richness and composition were similar among land-use types. Only one species (Warbling Vireo) experienced significant increases in abundance within campgrounds, whereas five species (Mourning Dove, Dusky Flycatcher, Black-billed Magpie, House Wren, and European Starling) responded positively in areas managed for grazing. Warbling Vireos generally placed their nests high in the canopy layer, which was unaffected by land-use activities (at least over the short-term) and relatively distant from recreationists. Most species with increased abundances in grazed lands are noted for using human-altered habitats (e.g., Saab 1999, Rodenhouse et al. 1995, Saab et al. 1995).

In a review of nine studies that evaluated avian responses to livestock grazing in riparian habitats (Saab et al. 1995), nearly half (46%) of 68 neotropical migrant landbirds decreased in abundance with cattle grazing, 9% increased with grazing, and 25% showed no clear response. Grouped either by nest placement or nest type, ground-nesting birds (including Veery and Fox Sparrow) were most negatively affected by livestock grazing, whereas canopy- and cavity-nesting species were least affected by grazing activities over the short-term. Results of this study were consistent with these findings.

I found that ground-nesting species were most susceptible to disturbances created by livestock grazing and were also most sensitive to fragmentation of riparian habitats, i.e. their relative abundances decreased with decreasing patch size. This is consistent with studies of habitat fragmentation in deciduous forests of eastern and mid-western United States (see Askins et al. 1990, Faaborg et al. 1995). Species that are more abundant on large fragments tend to be long-distance neotropical migrants rather than short-distance migrants or residents, generally nest on or near the ground, and use open rather than cavity nests (Whitcomb et al. 1981, Martin 1988, Faaborg et al. 1995). In this study, both Veery and Fox Sparrow were long-distance neotropical migrants that placed their open-cup nests (and foraged) on or near the ground. These ground nesters were probably more vulnerable to nest losses, and reductions in foraging habitat through the physical removal and damages to ground vegetation in grazed areas.

Factors affecting ground nesters in small fragments can include habitat alterations due to changes in microclimate conditions (cf. Faaborg et al.1995). Temperature and evaporation rates were higher next to openings than within continuous tropical forest (Lovejoy et al.1986). Such changes have been found to extend 30 to > 240 m into temperate forests of the Pacific Northwest United States (Chen et al. 1995).

Shrub nesters might be sensitive to changes resulting from livestock grazing and recreational activities, based on near significant ($p = 0.11$) decreases in their abundances as compared to unmanaged sites. Within grazed and recreation sites, microhabitats of shrub nesters were severely altered by significant reductions of shrub cover and densities, and increases in bare ground. By reducing foliage densities or opening dense patches of vegetation, cattle or campgrounds may increase nest losses by exposing concealed nests to predators (e.g., black-billed magpies) and allowing predator access. These results are consistent with regional trends. From 1968-1994 within the interior Columbia River Basin (which includes all of Idaho) species with decreasing populations tended to be those nesting in the shrub layer, whereas species with increasing populations tended to nest in tree canopies (Saab and Rich 1997). In forested habitats, songbirds that nest in shrubs generally experience the highest rates of nest predation (Martin 1993, Martin 1995), a factor that may be contributing to decreasing population trends within the region.

Mourning Dove, Yellow Warbler, and Song Sparrow are shrub-nesting species experiencing long-term population declines in the region (interior Columbia River Basin [Saab and Rich 1997]). Their abundances were significantly reduced with either local grazing, recreational activities, and/or small habitat fragments within my study area. Local land-use practices could be working in synergistic ways to cause widespread, regional declines within these species.

Canopy-nesting species tended to increase in grazed habitats as compared to recreation or unmanaged areas. Regionally, canopy nesters as a group experienced long-term population increases from 1968-1994 (Saab and Rich 1997). Few species in this group were affected by patch size along the South Fork. Additionally, the microhabitat feature of tree canopy coverage was similar among land uses, indicating that other factors were influencing patterns of habitat use by canopy nesters. A significant negative correlation was found between canopy nesters (which showed increases in grazed areas) and ground nesters (which showed decreases in grazed areas), suggesting that changes in interactions of the bird community could be affected by habitat modifications.

All cavity nesters were classified as nesting in the canopy and, when analyzed separately, they showed similar trends as canopy nesters. These results corroborate those of individual studies that examined short-term grazing effects on cavity nesters (see Saab et al.1995), and concluded that woodpeckers and other cavity-nesting species were relatively unaffected and sometimes increased in grazed habitats. Cavity-nesting birds place their nests in snags and dead limbs, and frequently forage in tree locations (bark) that are generally not used by cattle.

Relative abundances of Veeries and Fox Sparrows were reduced by half in recreation campgrounds compared to unmanaged sites, and both species were nearly absent from grazed areas. Fox Sparrows were associated with undeveloped sites when compared to campgrounds in riparian habitats of Utah (Blakesley and Reese 1988). In this Utah study, open-ground foragers such as American Robin and Gray Catbird were associated with campgrounds (although this was not statistically significant). These species were possibly attracted to food sources created by humans, whereas more wary species such as the Fox Sparrow avoided human activities (Garton et al.1977). Abundances of American Robins and Gray Catbirds in my study area appeared unaffected by recreational activities.

Veeries tended to use larger cottonwood patches (Saab 1999), particularly in recreation areas with campgrounds. Research in the Midwest and eastern United States has shown that Veeries are sensitive to reductions in sizes of forest patches, and avoid relatively small forest tracts (Robbins 1980, Robbins et al. 1989b, Herkert 1995). The Veery is experiencing significant population declines throughout the North American continent (Peterjohn et al. 1995). Habitat fragmentation, cattle grazing, and recreational activities within riparian systems could be factors contributing to their population declines in the western United States.

Five species, including Veeries, were unaffected by patch size in unmanaged areas with minimal use by humans or cattle, but showed significant area effects (increases in probability of occurrence with cottonwood forest area) in grazed and/or recreation sites. Perhaps larger patches of cottonwood riparian forests are required in disturbed areas (of relatively poor habitat quality) for these species to obtain all the resources needed for reproduction and survival. Small habitat patches disturbed by cattle or recreationists could be functioning as population "sinks," where reproduction does not compensate for adult mortality (Brown and Kodric-Brown 1977, Pulliam 1988). Alternatively, large patches might be acting as population "sources," where reproduction equals or exceeds adult mortality (Pulliam 1988).

MANAGEMENT IMPLICATIONS AND RESEARCH NEEDS

More than any other habitat in western North America, riparian woodlands are centers of high diversity and abundance of birds (Bock et al.1993a). Livestock grazing and recreation are concentrated within western riparian habitats, and many landbirds have responded negatively to these activities. Livestock grazing is the most widespread economic use of public lands in western North America (Platts 1991), while recreational activities continue to increase on shrinking land bases (Knight and Temple 1995). In the absence of effective management, these influences are likely to become more problematic for native plant and animal species.

Management practices often used to control recreational use of natural areas include collecting user fees, restricting visitor behavior and access, requiring permits based on specific qualifications, zoning, educating the public, limiting the number of visitors, and periodic closing (see van der Zande et al. 1984, Klein et al. 1995). Limiting visitor access and allowing the intensity of already busy areas to increase, rather than allowing visitor intensity to spread will likely reduce impacts to breeding birds (van der Zande et al. 1984). Along the South Fork of the

Snake River, the heaviest recreational use is during the breeding season, so periodic closing may not be an appropriate practice. Constraining the behavior of visitors is a viable management option because such things as noise, speed, and type of recreational activity elicit different responses from wildlife (Klein 1993). Aspects of these categories could be altered to minimize the impacts of recreationists. If noise and movement of recreationists could be reduced, there would be an increased likelihood of coexistence (Knight and Temple 1995). A lack of information on how human behavior affects wildlife has kept the usefulness of this coexistence strategy from being applied (Knight and Temple 1995).

The condition of riparian areas must be considered critically when implementing grazing systems, and, when practical, riparian woodlands should be managed separately from adjacent uplands (Platts 1991). Given their scarcity, fragility, and importance to landbirds and other wildlife, western riparian ecosystems should be excluded from livestock grazing wherever possible. Few bird species appear to benefit from grazing in these habitats, and those that do are not restricted to riparian communities (this study, Saab et al. 1995). Based on available information, when riparian systems are grazed, moderate use during late-fall and winter, or short-term use in spring, will be less damaging than continuous or growing-season grazing (Saab et al. 1995). Fall-winter grazing should be carefully controlled to ensure the maintenance of residual plant cover.

Degraded riparian habitats may require complete rest from livestock grazing to initiate the recovery process. The establishment of large protected areas (ca. 1000 ha) are needed to serve as references for comparison with managed sites (cf. Bock et al. 1993b). Four years after cattle removal from riparian habitat in Arizona, understory vegetation and Neotropical migrants showed dramatic increases in abundance (Krueper 1993). In systems requiring long-term rest, the necessary period will be highly variable depending upon the extent of damage and growth rate of regenerating plant species (Clary and Webster 1989). Damaged riparian areas should be rehabilitated by revegetating with native species.

Riparian habitats of arid land western North America have unique features among forests (i.e., linear, narrow shapes with large amounts of edge) and are often naturally fragmented. Yet some species could be characterized as large patch, interior specialists (e.g.,Veery), and others are clearly edge specialists (e.g., Song Sparrow). Thus, management considerations should include conservation of both large (> 10 ha in cottonwood woodlands) and small patches, although small patch/edge habitats usually are not limiting. Conservation of large patches is particularly important where riparian forests are managed for grazing and recreation. Some species apparently need larger patches of breeding habitat in areas with these disturbances.

Further research is needed to improve our understanding of the relationships between landbirds and land-use practices in riparian ecosystems. Studies designed to evaluate the types, timing, and intensity of grazing and recreational activities are needed to determine the degree of intolerance or habituation of the birds. No information is available on the synergistic effects of grazing and recreation, and this is critically needed because these land-use activities frequently occur together in the western United States. If grazing and recreational disturbance are responsible for individuals leaving an area, we need to know where they go and what habitats they use, and the relationships of riparian habitats to other parts of the landscape that support riparian birds during the breeding season. Avian abundance data may not always reflect habitat suitability (Van Horne 1983). Long-term studies on reproductive success, survivorship, and population persistence are needed in riparian habitats under different management regimes.

LANDSCAPE INFLUENCES AND MANAGEMENT IMPLICATIONS

Information presented in the previous sections was part of a multi-study project to evaluate the effects of smaller-scale management practices and larger-scale landscape patterns on habitat relationships of breeding birds in cottonwood forests (Saab 1996). This section is summarized from Saab (1999) and provides a synopsis on the relative importance of landscape patterns to habitat use by breeding birds.

A hierarchical approach was used to examine habitat use at three spatial scales: microhabitat (local vegetation characteristics), macrohabitat (cottonwood forest patch characteristics), and landscape (composition and patterning of surrounding [matrix] vegetation types and land uses). A series of predictions regarding distributions of 32 species were addressed that incorporated the different spatial scales. The surrounding landscape changed from a valley surrounded by mountains on the upstream end of the study area, a narrow canyon adjacent to natural upland vegetation in the middle section, to a wide, open floodplain dominated by agriculture on the downstream end. The best predictors of high species richness of the native avifauna were: (1) natural and heterogeneous landscapes, (2) large cottonwood patches, (3) close proximity to other cottonwood patches, and (4) microhabitats with relatively open canopies. The most frequent significant predictor of species occurrence was the landscape component - increases in upland natural vegetation with decreases in agriculture. Both habitat interior and edge specialists were found in arid land, cottonwood riparian forests that are linear in nature with large amounts of edge. Nest predators (black-billed magpie and American crow), brood parasites (brown-headed cowbird), and exotic species (European starling) responded positively to human-altered landscapes. Landscape patterns were the primary influence on distribution and occurrence of most bird species, while macrohabitat and microhabitat were of secondary importance. Thus, surrounding landscape (matrix) features should be a primary consideration for managing riparian habitats and selecting riparian reserve areas.

Land acquisition and maintenance of large cottonwood patches surrounded by natural landscapes should take precedence over conserving large patches surrounded by agriculture if maintaining high species richness of native birds is a management objective. Conservation of contiguous patches of cottonwood forest adjacent to palustrine wetlands is also desirable for many individual species and for maintenance of species richness. Both large and naturally small fragments of riparian habitat are needed for conservation of interior and edge specialists. Small patches, generally are not limiting in arid-land riparian habitats, but those that exist should be conserved for bird species associated with edge habitats. Management objectives for natural landscapes should consider controlling residential growth to reduce the likelihood of avian nest predators (i.e., crows and magpies) and exotic species (i.e., starlings). Among microhabitat characteristics, a relatively open cottonwood forest canopy was the most important predictor of high species richness and of occurrence for several species. This microhabitat feature may reflect pre-dam conditions, when natural flooding disturbances created more patchiness in the mature forest canopy interspersed with younger cottonwood stands (cf. Merigliano 1996). Flood control can greatly alter riparian plant communities by increasing cover of plant species that would otherwise be removed by flood scour, causing plant desiccation, reduced growth, competitive exclusion, ineffective seed dispersal, or failure of seedling establishment (see Poff et al. 1997). The magnitude and timing of peak flows should approximate pre-dam conditions for the long-term maintenance of cottonwood forests (Rood and Heinze-Milne 1989, Johnson 1992, Merigliano 1996) and the associated bird community.

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APPENDIX.1. Species used for data analysis and recorded within point count circles during 1991-1994, May-July along the South Fork Snake River in southeastern Idaho. Letters in parentheses after the common name indicate migratory status: L = long-distance neotropical migrant; S = short-distance neotropical migrant; R = resident. The letter "C" in parentheses after the scientific name indicates that the species nests in tree cavities; all other species are open-cup nesters. Total number of patches is the number of cottonwood patches from a total of 57 in which a species was recorded.

Common name	Acronym	Scientific name	Nest Layer[a]	Total No. Patches
American Kestrel(S)	AMKE	*Falco sparverius(C)*	CA	24
Mourning Dove(L)	MODO	*Zenaida macroura*	SH	50
Yellow-billed Cuckoo(L)	YBCU	*Coccyzus americanus*	CA	5
Red-naped Sapsucker(L)	RNSA	*Sphyrapicus nuchalis(C)*	CA	43
Hairy Woodpecker(R)	HAWO	*Picoides villosus(C)*	CA	10
Downy Woodpecker(R)	DOWO	*Picoides pubescens(C)*	CA	38
Northern Flicker(S)	NOFL	*Colaptes auratus(C)*	CA	50
Eastern Kingbird(L)	EAKI	*Tyrannus tyrannus*	CA	12
Western Wood-pewee(L)	WWPE	*Contopus sordidulus*	CA	40
Dusky Flycatcher(L)	DUFL	*Empidonax oberholseri*	SH	26
Black-billed Magpie(R)	BBMA	*Pica pica*	SH	46
American Crow(R)	AMCR	*Corvus brachyrhynchos*	CA	27
Black-capped Chickadee(R)	BCCH	*Parus atricaphillus(C)*	CA	51
House Wren(L)	HOWR	*Troglodytes aedon(C)*	CA	39
Gray Catbird(L)	GRCA	*Dumetella carolinensis*	SH	37
American Robin(S)	AMRO	*Turdus migratorius*	CA	57
Veery(L)	VEER	*Catharus fuscenscens*	GR	34
Cedar Waxwing(S)	CEWA	*Bombycilla cedorum*	SH	43
European Starling(R)	EUST	*Sturnus vulgaris(C)*	CA	44
Warbling Vireo(L)	WAVI	*Vireo gilvus*	CA	42
Red-eyed Vireo(L)	REVI	*Vireo olivaceous*	CA	11
Yellow Warbler(L)	YEWA	*Dendroica petechia*	SH	57
Yellow-rumped Warbler(S)	YRWA	*Dendroica coronata*	CA	32
MacGillivray's Warbler(L)	MGWA	*Oporornis tolmiei*	SH	26
Yellow-breasted Chat(L)	YBCH	*Icteria virens*	SH	12
Western Tanager(L)	WETA	*Piranga ludoviciana*	CA	27
Black-headed Grosbeak(L)	BHGR	*Pheucticus melanocephalus*	SH	46
Lazuli Bunting(L)	LZBU	*Passerina amoena*	SH	34
Northern Oriole(L)	NOOR	*Icterus glabula*	SH	54
Brown-headed Cowbird(S)	BHCO	*Molothrus ater*	SH	50
Cassin's Finch(S)	CAFI	*Carpodacus cassinii*	CA	19
American Goldfinch(S)	AMGO	*Carduelis tristis*	CA	56

[a] Nest Layer abbreviations: CA=subcanopy/canopy nesting species; GR=ground-nesting species; SH=shrub-nesting species based on characteristics described by Ehrlich et al. (1988), Martin (1993), and known nest locations within the study area (Saab, unpublished data).

APPENDIX 2. All bird species recorded within point count circles during 1991-1994 breeding seasons along the South Fork Snake River in southeastern Idaho. Species whose names are in bold (listed at the end) were observed outside of point count surveys or during surveys conducted by Whitfield and Maj (1998).

Common Name	Scientific Name	Common Name	Scientific Name
Double-crested Cormorant	*Phalacrocorax auritus*	Osprey	*Pandion haliaetus*
Great Blue Heron	*Ardea herodias*	Golden Eagle	*Aquila chrysaetos*
Sandhill Crane	*Grus canadensis*	Bald Eagle	*Haliaeetus leucocephalus*
Canada Goose	*Branta canadensis*	Red-tailed Hawk	*Buteo jamaicensis*
Mallard	*Anas platyrhynchos*	American Kestrel	*Falco sparverius*
Common Merganser	*Mergus merganser*	Northern Goshawk	*Accipiter gentilis*
Sora	*Porzana carolina*	Cooper's Hawk	*Accipiter cooperii*
Spotted Sandpiper	*Actitis macularia*	Sharp-shinned Hawk	*Accipiter striatus*
Killdeer	*Charadrius vociferus*	Ruffed Grouse	*Bonasa umbellus*
Red-necked Phalarope	*Phalaropus lobatus*	Rock Dove	*Columba livia*
Common Snipe	*Gallinago gallinago*	Mourning Dove	*Zenaida macroura*
Turkey Vulture	*Cathartes aura*	White-throated Swift	*Aeronautes saxatilis*

APPENDIX 2. Continued.

Common Name	Scientific Name	Common Name	Scientific Name
Yellow-billed Cuckoo	*Coccyzus americanus*	Willow Flycatcher	*Empidonax traillii*
Black-chinned Hummingbird	*Archilochus alexandri*	Dusky Flycatcher	*Empidonax oberholseri*
Calliope Hummingbird	*Stellula calliope*	*Empidonax* sp.	*Empidonax* sp.
Belted Kingfisher	*Ceryle alcyon*	Tree Swallow	*Tachycineta bicolor*
Great Horned Owl	*Bubo virginianus*	Violet-green Swallow	*Tachycineta thalassina*
Common Nighthawk	*Chordeiles minor*	Northern Rough-winged Swallow	*Stelgidopteryx serripennis*
Red-naped Sapsucker	*Sphyrapicus nuchalis*	Bank Swallow	*Riparia riparia*
Hairy Woodpecker	*Picoides villosus*	Cliff Swallow	*Hirundo pyrrhonota*
Downy Woodpecker	*Picoides pubescens*	Barn Swallow	*Hirundo rustica*
Northern Flicker	*Colaptes auratus*	Black-billed Magpie	*Pica pica*
Eastern Kingbird	*Tyrannus tyrannus*	Common Raven	*Corvus corax*
Olive-sided Flycatcher	*Contopus borealis*	American Crow	*Corvus brachyrhynchos*
Western Wood-pewee	*Contopus sordidulus*	Mountain Chickadee	*Parus gambeli*

APPENDIX 2. Continued.

Common Name	Scientific Name	Common Name	Scientific Name
Black-capped Chickadee	*Parus atricaphillus*	Cedar Waxwing	*Bombycilla cedorum*
Red-breasted Nuthatch	*Sitta canadensis*	European Starling	*Sturnus vulgaris*
Rufous Hummingbird	*Selasphorus rufus*	Solitary Vireo	*Vireo solitarius*
Broad-tailed Hummingbird	*Selasphorus platycercus*	Warbling Vireo	*Vireo gilvus*
House Wren	*Troglodytes aedon*	Red-eyed Vireo	*Vireo olivaceus*
Ruby-crowned Kinglet	*Regulus calendula*	Yellow Warbler	*Dendroica petechia*
Gray Catbird	*Dumetella carolinensis*	Yellow-rumped Warbler	*Dendroica coronata*
Townsend's Solitaire	*Myadestes townsendi*	Black-throated Gray Warbler	*Dendroica nigrescens*
American Robin	*Turdus migratorius*	MacGillivray's Warbler	*Oporornis tolmiei*
Swainson's Thrush	*Catharus ustulatus*	Orange-crowned Warbler	*Vermivora celata*
Veery	*Catharus fuscenscens*	Yellow-breasted Chat	*Icteria virens*
Hermit Thrush	*Catharus guttatus*	Common Yellowthroat	*Geothlypis trichas*
American Dipper	*Cinclus mexicanus*	Western Tanager	*Piranga ludoviciana*

APPENDIX 2. Continued.

Common Name	Scientific Name	Common Name	Scientific Name
Black-headed Grosbeak	*Pheucticus melanocephalus*	House Sparrow	*Passer domesticus*
Lazuli Bunting	*Passerina amoena*	Cassin's Finch	*Carpodacus cassinii*
Green-tailed Towhee	*Pipilo chlorurus*	Pine Siskin	*Carduelis spinus*
Chipping Sparrow	*Spizella passerina*	American Goldfinch	*Carduelis tristis*
Dark-eyed Junco	*Junco hyemalis*	Red Crossbill	*Losiz curvirostra*
Western Meadowlark	*Strunella neglecta*	White-crowned Sparrow	*Zonotrichia leucophrys*
Red-winged Blackbird	*Agelaius phoeniceus*	Fox Sparrow	*Passerella iliaca*
Yellow-headed Blackbird	*Xanthocephalus xanthocephalus*	Song Sparrow	*Melospiza melodia*
Brewer's Blackbird	*Euphagus cyanocephalus*	Evening Grosbeak	*Coccothraustes vespertinus*
Common Grackle	*Quiscalus quiscala*	**Swainson's Hawk**	***Buteo swainsoni***
Bullock's Oriole	*Icterus bullockii*	**Northern Harrier**	***Circus cyaneus***
Brown-headed Cowbird	*Molothrus ater*	**Peregrine Falcon**	***Falco peregrinus***

APPENDIX 2. Continued.

Common Name	Scientific Name	Common Name	Scientific Name
Prairie Falcon	***Falco mexicanus***		
Northern Saw-whet Owl	***Aegolius acadicus***		
Northern Pygmy-Owl	***Glaucidium gnoma***		
Western Screech Owl	***Otus kennicottii***		
Flammulated Owl	***Otus flammeolus***		

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