DISTRIBUTION OF SONGBIRDS IN RIPARIAN FORESTS OF CENTRAL MAINE

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DISTRIBUTION OF SONGBIRDS IN RIPARIAN FORESTS OF CENTRAL MAINE

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I investigated the relative influences of distance from streams and vegetation structure on distributions of songbirds in riparian forests of Maine in 1989-1990. I surveyed 24 riparian sites in 1989 and 18 sites in 1990, classifying each site as either upland or floodplain riparian forest. Upland forests (21 sites) were well-drained, coniferous or deciduous forests, adjacent to a stream or river, and were similar in vegetation structure to non-riparian forests. Floodplain forests (8 sites) were poorly drained, deciduous forests, often with standing water. At each site, I censused breeding birds and measured 14 vegetation variables along transects at 6 distances from streams: 25 m, 75 m, 125 m, 175 m, 225 m, 275 m. The distributions of songbirds among these 6 distances were compared between upland and floodplain forest types, with the null hypothesis that species richness and numbers of songbirds would not vary with increasing distance from streams (i.e. no "riparian effect").

Patterns of songbird distribution differed between upland and floodplain forests. Species richness showed no strong trends with distance in upland forests.

In floodplain forests, species richness decreased from 25 m to 125 m from streams; >125 m from streams, no differences were evident in species richness. I detected similar distribution patterns relative to distance from streams for total individuals in each forest type.

Forty-eight percent of the songbird community occurred within both forest types. All songbird species detected in my study occur in other habitats in central Maine. Interior species dominated in upland forests, while edge species dominated in floodplain forests. Number of individuals of edge species decreased with distance from streams in floodplain forests, but were independent of distance in upland forests. Number of individuals of interior species were independent of distance in both forest types.

Upland forests, dominated by conifers, large trees, and closed canopy, formed abrupt interfaces with streams, and vegetation did not vary with distance from streams. Floodplain forests, dominated by more open canopy, smaller trees, and many deciduous shrubs, provided a transitional habitat, with conifers increasing ≥175 m from streams. Songbirds were significantly associated with variations in vegetation among sites and between the 2 forest types.

I detected no riparian effect for species richness or total individuals of songbirds in upland forests of central Maine. This directly contrasts conditions in the southwestern U.S., where the presence of riparian forests significantly increases regional songbird diversity. A riparian effect was evident in floodplain forests in my study; however, the increase in songbirds was primarily generalist species. My study neither adequately addressed reproductive or foraging success of songbirds, nor life requirements of non-passerine bird species in riparian forests of central Maine.

Using prerecorded bird songs representing 2 song volumes and 4 song patterns, I conducted a test near 4 "loud" streams and 4 "quiet" streams to determine if noise created by streams affected my ability to detect singing male birds at 3 distances: 25 m, 75 m, and 125 m. I heard all songs played at 25 m from quiet streams, and at distances $\geq$75 m from both quiet and loud streams. I heard >95% of songs played at 25 m from loud streams. Results of this test indicated that I probably missed few, if any, birds during my censuses because of stream noise.

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By

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A THESIS

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CHAPTER I

GENERAL INTRODUCTION

Riparian zones have recently gained recognition for their importance in maintaining the integrity of aquatic ecosystems and for providing habitat that is disproportionately used by wildlife. However, riparian zones are especially susceptible to the effects of recreation, grazing, flood control, development, and timber harvesting because of their linearity, relatively small area, and proximity to water (Thomas et al. 1979, Knopf et al. 1988). Alternative uses for riparian zones, therefore, create decisions with a high potential for conflict between wildlife resources and human users (Knopf et al. 1988).

Riparian ecosystems are generally regarded as relatively mesic plant communities occurring as a transitional zone between aquatic and more xeric upland sites (Ohmart and Anderson 1986, Knopf et al. 1988). This view originated in the western United States, where the moist conditions of riparian zones provide a sharp contrast to the water-limited upland (Johnson and Lowe 1985). Although less than 1% of the western landscape of the United States is covered by riparian vegetation (Knopf et al. 1988), these habitats maintain substantially greater productivity and diversity of wildlife species than upland habitats (Brinson et al. 1981, Szaro and Jakle 1985, Knopf 1986). The greater structural diversity and moisture provided by riparian vegetation in western regions provides habitat for bird species that could not otherwise breed in xeric uplands (Szaro and Jakle 1985, Knopf 1986). For example, 35-50% of bird species in the southwestern U.S. are restricted to riparian zones (Johnson et al. 1977, review by Brinson et al. 1981, Szaro and Jakle 1985). Thus, loss of riparian habitat in the Southwest would significantly decrease regional bird species richness. In the Great Plains, development of riparian vegetation has provided dispersal corridors for bird species that were previously restricted by the expanse of open grasslands (Knopf 1986, Finch 1989). Seventy-one percent of species using riparian zones in eastern Colorado had continental or eastern distributions, whereas only 4% were native to the Great Plains (Knopf 1986). Bird communities from riparian habitats also contribute substantially to increasing species richness in upland habitats (Gaines 1974, Johnson et al. 1977, Szaro and Jakle 1985). For example, 75% of species using riparian zones in Arizona were also detected in the adjacent scrub habitat (Szaro and Jakle 1985). The presence of riparian bird species in adjacent desert upland nearly doubled species richness compared to a similar desert habitat not adjacent to a riparian zone (Szaro and Jakle 1985). Because of this significant productivity, the importance of western riparian habitats to wildlife has been the focus of much research and has become a central issue in managing public lands.

For about the last 10 years, research documenting the value of riparian habitats to both aquatic and terrestrial ecosystems has proliferated, and numerous symposia have focused on conserving and managing riparian zones. However, this information is biased toward western riparian zones. Of 44 studies of wildlife and riparian zones reported in the United States between 1978-1988, 35 (80%) were conducted west of the Mississippi River (Hooper 1989). As an extreme comparison, 28 studies were conducted in Arizona, compared to 3 in Maine.

Why this imbalance? In contrast to the west, water is more evenly distribted in Maine, making riparian zones less ecologically distinct from adjacent areas (Brown et al. 1978, Johnson and Lowe 1985). Habitats bordering rivers and lakes in Maine range from water-saturated peatlands to relatively well-drained forests that often extend many kilometers from the water. Most research on riparian habitats conducted in the northeastern U.S. and Canada relates to the influences of riparian vegetation on water quality and fisheries (Hooper 1989). Riparian vegetation moderates water temperatures of streams, which is critical in maintaining many fish species and other aquatic organisms (Barton et al. 1985, Moring et al. 1985, Moring and Garmon 1986). One 1982 study on the East Branch of the Piscataquis River in Maine reported that water temperatures were significantly higher following logging along the river, with temperature maxima exceeding 30° C (86° F) on several dates in the summer (Garmon 1984). Generally, temperatures $\leq$20°-24° are needed to maintain trout species (Barton et al. 1985). Riparian vegetation also acts as a filter system, controlling runoff from the adjacent lands and reducing erosion of the stream banks (Moring et al. 1985, Moring and Garmon 1986). Finally, riparian vegetation can input nutrients and invertebrates into the stream system for fish food, and can provide overhanging vegetation as cover for fish and other aquatic animals (Moring et al. 1985, Moring and Garmon 1986). Approximately 30-45 m of riparian vegetation is recommended to maintain water quality and fisheries (Barton et al. 1985, Howard and Allen 1988, Jones et al. 1989).

Many states in the northeastern U.S. have recognized the importance of these functions of riparian vegetation for water quality and fisheries, and have developed regulations for managing riparian habitats (Small and Johnson 1986). In Maine, regulations in the unorganized townships currently require a 76 m (250 ft) buffer strip along streams and great ponds, with limited timber harvesting allowed within the buffer strip (Maine Land Use Regulation Commission 1986). Although riparian zones in Maine are also considered important to terrestrial wildlife (Small and Johnson 1986, Jones et al. 1989), data supporting this conclusion have often come from studies conducted in the western U.S. (Hooper 1989). Results from western studies are difficult to apply to Maine because hydrological conditions in Maine are so different from those in the west (Johnson and Lowe 1985). Wildlife managers and land-use planners in Maine need studies conducted under local hydrological conditions to make effective decisions concerning the protection and management of riparian habitats for wildlife.

Studies of use of northeastern riparian zones by songbirds have focused on the influence of human disturbance between disturbed and undisturbed riparian habitats, or on sites along estuarine rivers. Johnson and Brown (1990) reported lower breeding bird density, diversity, and richness in a harvested lakeshore buffer zone compared to an undisturbed lakeshore in Maine. In contrast, on Prince Edward Island, Canada, bird species richness was higher in managed (cut) than unmanaged sites along the Montague River, with substantial differences in species composition between the 2 types of sites (S. Makepeace, unpubl. report, 1989 Avian Surveys in Eastern Prince Edward Island, Montague Watershed Project, 1989). No significant difference in relative abundance of birds occurred between site types. Small (1986) reported no increase in bird species richness or abundance $\leq$30 m from an abrupt forest-river edge compared to 30-90 m into the forest at 2 estuarine sites in southern Maine.

I examined songbird distributions in relatively undisturbed forest stands adjacent to inland streams and rivers in Maine during 1989 and 1990 and assessed the influences of distance from streams and habitat characteristics on songbird use of these riparian forests (Chapter II). For this thesis, I defined riparian forests as any forest adjacent to a stream or river. I recognized 2 different classes of riparian forests. Floodplain forests were flat, poorly drained, deciduous riparian forests, with standing water at least some part of the year; upland forests were moderately to steeply sloped, well-drained, coniferous, deciduous, or mixed-wood riparian forests similar in vegetation structure and composition to non-riparian forests. My objective was to determine the relationships between distance from streams and number of songbird species, total individuals of all songbirds, and number of individuals of each songbird species. To facilitate this objective, I examined associations between habitat characteristics of riparian forests and distributions of songbirds, assessing effects of both general forest type (floodplain or upland) and microhabitat variation. I also tested whether background noise created by streams decreased my probability of detecting bird songs to ensure that surveys of singing male birds were an appropriate index to distributions of songbirds relative to riparian edges (Chapter III).

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CHAPTER II

DISTRIBUTION OF SONGBIRDS IN RIPARIAN FORESTS OF MAINE

Riparian zones are considered important for maintaining the integrity of aquatic ecosystems and for providing habitat that is disproportionately valuable for wildlife. In arid regions of the United States, moist conditions of riparian habitats provide a sharp contrast to the water-limited upland (Johnson and Lowe 1985). Although less than 1% of the western landscape is covered by riparian vegetation (Knopf et al. 1988), these habitats substantially increase the diversity of species in western habitats (Szaro and Jakle 1985, Knopf 1986). For example, loss of the riparian component in the southwestern states could potentially result in the loss of up to 47% of the bird species breeding in that region (Johnson et al. 1977). In arid regions, riparian zones provide habitat for bird species that could not otherwise nest in the upland habitats such as desert scrub (review by Brinson et al. 1981; Szaro and Jakle 1985; Hunter et al. 1987) or grasslands of the Great Plains (Knopf 1986, Finch 1989). Consequently, the importance of western riparian habitats to wildlife has been the focus of much research and has become a central issue in managing public lands.

In contrast to the west, water is more evenly distributed in Maine, making riparian zones less ecologically distinct from adjacent areas (Brown et al. 1978, Johnson and Lowe 1985). Habitats bordering rivers and lakes in Maine range from water-saturated peatlands to relatively well-drained upland forests, which often extend many kilometers from the river. Most research on riparian habitats conducted in the northeastern U.S. and Canada relates to the influence of riparian vegetation on water quality and fisheries (e.g. Garmon 1984, Barton et al. 1985, Moring et al. 1985, Moring and Garmon 1986). Although riparian zones in Maine are also considered important to terrestrial wildlife (Small and Johnson 1986, Jones et al. 1988), data supporting this conclusion have often come from studies conducted in the southwestern U.S. (Hooper 1989). Wildlife managers in Maine need studies conducted under local hydrological conditions to make effective decisions concerning the protection and management of riparian habitats for wildlife.

Studies of use of eastern riparian zones by songbirds have focused on the influence of human distrubance between disturbed and undisturbed riparian habitats (Stauffer and Best 1980, Johnson and Brown 1990) or on sites along estuarine rivers (Small 1986). No studies in Maine have extensively examined use of undisturbed riparian forests by songbirds. I examined songbird distributions in relatively undisturbed forest stands adjacent to inland streams and rivers in Maine during 1989 and 1990, and assessed the influences of distance from streams and habitat characteristics on songbird use of these riparian forests. For this thesis, I defined riparian forests as any forest adjacent to a stream or river. My objective was to determine the relationships between distance from streams and number of songbird species, total individuals of all songbirds, and number of individuals of each songbird species. To facilitate this objective, I examined associations between habitat characteristics of riparian forests and distributions of songbirds, assessing the effects of general forest type and microhabitat variation.

STUDY AREAS

The original protocol for this study involved an intensive survey of 3 riparian sites (2 streams, 1 pond). However, the 3 original sites contained substantial habitat patchiness, and a pilot study in 1987 indicated that this patchiness may confound distributions of birds within these sites (Appendix B). Furthermore, in 1988, I could not locate sites with relatively homogeneous vegetation that were large enough to provide adequate replicability for an intensive study (Appendix B). Therefore, in 1989, I conducted an extensive survey of 24 sites throughout Maine, to obtain a representative sample of the types of forested riparian habitats that land-use planners must consider. I classified each site into 1 of 2 general types of riparian forest, floodplain or upland, based on slope, hydrological conditions, and composition of the vegetation. I defined floodplain forests as flat, poorly-drained, deciduous riparian forests, with standing water at least some part of the year. I defined upland forests as moderately to steeply sloped, well-drained, coniferous or deciduous riparian forests, similar in vegetation structure and composition to non-riparian forests. In 1989, 19 sites were upland forests, 3 sites were floodplain forests, and 2 sites had a floodplain component $\leq$75 m from the stream (Table 1; Fig. 1). In 1990, I resurveyed 13 of these sites, and added 5 floodplain sites to the survey (Table 1; Fig. 1).

Table 1. Stream names, locations, forest types, and census years for 29 study sites of riparian forests, Maine, 1989-1990.

Map no.[a]	Site code	Stream name	Location		Forest type	Year censused	
			township	county		1989	1990
1	SAC1	Saco River	Fryeburg	Oxford	floodplain		X
2	SAC2	Saco River	Fryeburg	Oxford	floodplain		X
3	DTR	Duck Trap River	Lincolnville	Waldo	upland	X	X
4	SER	Sebasticook River	Burnham	Waldo	upland	X	X
5	MS	Marsh Stream	Frankfort	Waldo	upland	X	X
6	SBC	South Branch Carrabasset River	Carrabasset Valley	Franklin	upland	X	
7	WS2	Wesserunsett Stream	Skowhegan	Somerset	upland	X	
8	WS1	Wesserunsett Stream	Cornville	Somerset	upland	X	X
9	EBS1	East Branch Sebasticook River	Plymouth	Penobscot	upland	X	
10	EBS2	East Branch Sebasticook River	Newport	Penobscot	upland	X	
11	SOS2	Souadabscook Stream	Hampden	Penobscot	floodplain	X	X
12	SOS1	Souadabscook Stream	Hampden	Penobscot	upland	X	X
13	BLS	Blackman Stream	Bradley	Penobscot	upland	X	
14	WBU	West Branch Union River	Amherst	Hancock	upland	X	X
15	MBU1	Middle Branch Union River	Aurora	Hancock	upland	X	X
16	MBU2	Middle Branch Union River	Aurora	Hancock	upland	X	
17	SUS	Sunkhaze Stream	Milford	Penobscot	upland[b]	X	

Table 1. Continued.

Map no.	Site code	Stream name	Location		Forest type	Year censused	
			township	county		1989	1990
18	PUS	Pushaw Stream	Old Town	Penobscot	upland[b]	X	
19	PUS2	Pushaw Stream	Old Town	Penobscot	floodplain		X
20	BS2	Birch Stream	Old Town	Penobscot	floodplain	X	X
21	BS3	Birch Stream	Argyle	Penobscot	floodplain	X	X
22	BS1	Birch Stream	Argyle	Penobscot	upland	X	
23	OS	Olamon Stream	Greenbush	Penobscot	upland	X	
24	PAS	Passadumkeag River	Lowell	Penobscot	upland	X	X
25	MAD1	Madagascal Stream	Grand Falls	Penobscot	floodplain		X
26	MAD2	Madagascal Stream	Grand Falls	Penobscot	floodplain		X
27	WBP	West Branch Piscataquis River	Blanchard	Piscataquis	upland	X	X
28	NS	Nahmakanta Stream	T1 R11 WELS	Piscataquis	upland	X	X
29	RS	Rainbow Stream	Rainbow	Piscataquis	upland	X	

[a] Map No. refers to numbered locations on Fig. 1.

[b] floodplain component $\leq$75 m from streams; upland forest >75 m from streams.

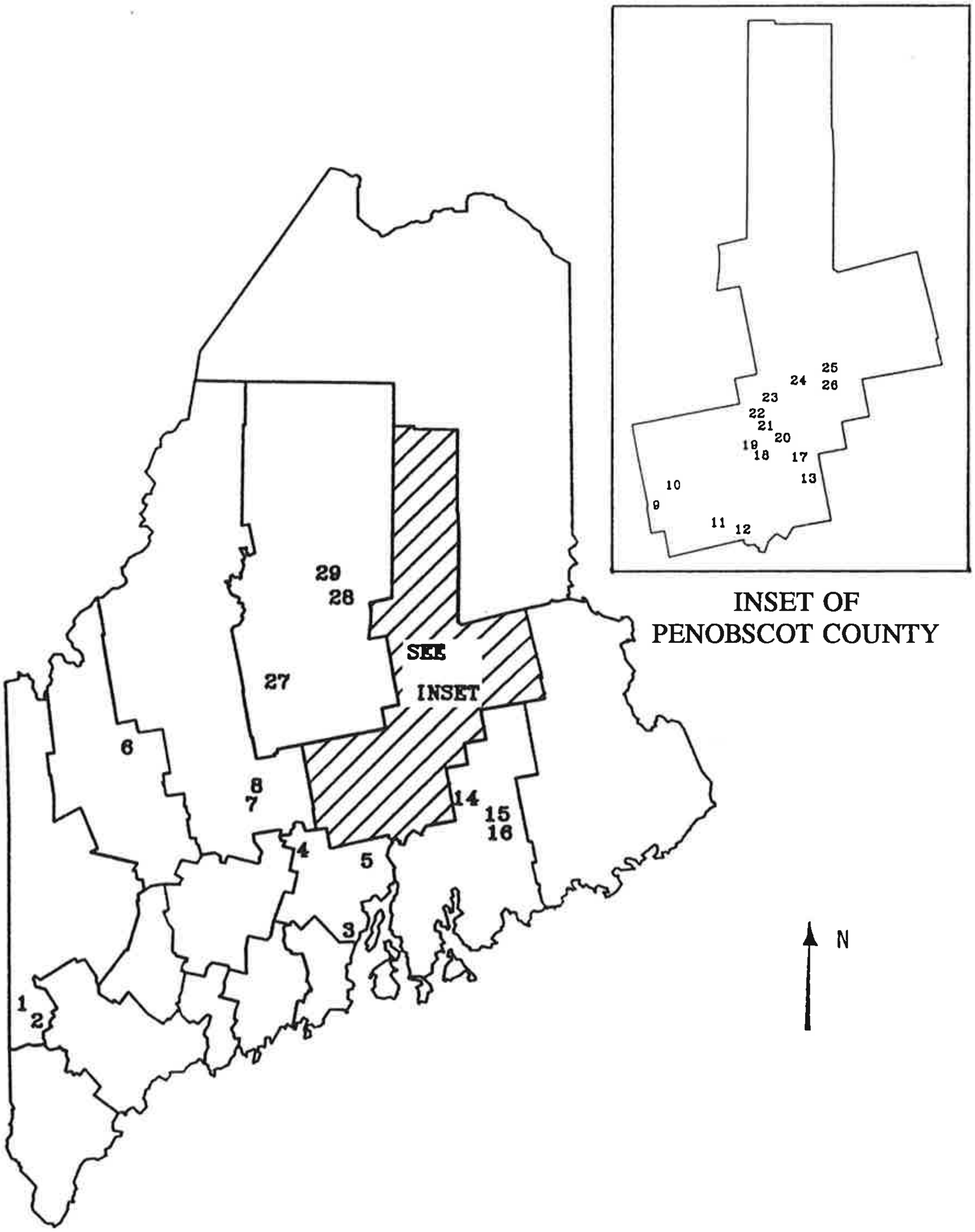

Figure 1. Locations of 29 riparian forest sites in Maine where birds were censused in 1989 or 1990. Numbers refer to Table 1, which identifies name, year(s) censused, and forest type of each site.

Coniferous upland forests were generally dominated by hemlock (Tsuga canadensis), balsam fir (Abies balsamea), and red spruce (Picea rubens). Deciduous upland forests were dominated by red maple (Acer rubrum), white birch (Betula papyrifera), sugar maple (Acer saccharum), and American beech (Fagus grandifolia). Mixed-wood forests had combinations of these species. Floodplain forests were dominated by silver maple (Acer saccharinum) and red maple. I tried to select sites that had relatively homogeneous canopy composition (coniferous, mixed-wood, or deciduous) within the site; however, many sites contained some habitat patchiness. All streams were 10-30 m wide, providing a distinct gap in the forest canopy, thereby making the stream a conspicuous feature of the local landscape.

METHODS

Distributions of Songbirds

Breeding bird surveys.--I conducted surveys of breeding birds from 29 May-7 July 1989 and 29 May-27 June 1990. Ten of 24 sites in 1989 and all 18 sites in 1990 were each censused once (Table 1). In 1989, the remaining 14 sites were censused twice during the season to test for consistency between multiple censuses within a year. I did not detect differences in species richness or number of songbirds between 2 censuses at these 14 sites (Appendix C), and further analyses for this thesis are therefore based on only the first census. Hutto et al. (1986) recommended fewer censuses at more plots to increase the number of independent samples in a study. By censusing each site only once, I was able to include more sites in the study, thereby increasing my sample of riparian forests in Maine. Twelve of the 18 sites censused in 1990 were sites I had randomly selected from the original 24 sites to test for consistency in censuses between years.

I censused 1 or 2 sites each morning between 0500 and 0930. Censuses were not conducted on rainy or windy (>16 kph) mornings. At each site I sampled along 3 transects, 50 m apart, set perpendicular to the stream and extending 300 m into the forest. I censused for 3 minutes at 50 m intervals along the transects, beginning 25 m from the stream. This design provided 18 sampling points per site, with 3 sampling points at each of 6 distances from the stream (25 m, 75 m, 125 m, 175 m, 225 m, 275 m) at each site (4.5-ha sampling area). I started at different sampling points for each census to avoid systematic biases that could occur when consistently censusing the same sampling point at the same time of day (Robbins 1981, Skirvin 1981). All birds seen or heard within a 25 m-radius circle were recorded (Hutto et al. 1986). Waide and Narins (1988) recommended <50 m fixed-radius circles for censusing breeding birds to reduce the effects of attenuation of bird songs by forest vegetation while allowing for adequate sample sizes within a censusing plot. I also recorded all bird species detected during a 3-minute listening period within a 50-m strip at the forest-stream interface (distance = 0 m to mid-stream) centered at each transect. I did not use these latter counts in subsequent analyses, but used them to expand the list of species occupying riparian edges.

Censuses were used to determine distributions of bird species relative to the 6 distances from streams. I calculated species richness, total individuals for all species, and number of individuals for each species by summing the respective totals from the 3 transects (25 m-radius plots) for each distance at each site. I classified species as "edge" or "interior" based on their habitat use reported in previous studies (Godfrey 1986, Small 1986, Elliot 1987, Robbins et al. 1989). Edge species included those species commonly associated (e.g. highest abundances or densities) with the transition zones between 2 habitats. Interior species included those that are generally forest-area sensitive, or are associated with interior of forests. Many species from my censuses did not fit either category. Therefore, I selected the 7 species from each category that were most abundant in my censuses for analyses. Edge species comprised American robin, least flycatcher, common yellowthroat, blue jay, white-throated sparrow, yellow warbler, and chestnut-sided warbler. Interior species comprised ovenbird, golden-crowned kinglet, veery, hermit thrush, northern parula, black-throated blue warbler, and solitary vireo. Appendix A lists common and scientific names of all bird species identified over the 2 years of the study. I calculated abundances for edge and interior species at each distance by summing over the 3 transects the number of individuals of the respective 7 species.

The primary assumption in calculating density is that all birds are detected during a census, and this is not valid for many species (Mayfield 1981, Hutto et al. 1986). Hence, I considered each bird parameter in my study a relative value rather than calculating absolute bird densities. A test with prerecorded bird songs (Chapter III) indicated that background noise caused by loud, rushing rivers did not decrease bird counts 25 m from rivers.

Statistical Analyses.--I tested for consistency in the distributions of songbirds over distance from streams for the 12 sites sampled both in 1989 and 1990 using repeated measures analysis of variance (ANOVA) for profile analysis (Timm 1975:237-250, SAS 1985:478-483). This analysis is a multivariate ANOVA technique for counts taken on the same experimental unit (in this case, the same 12 sites). The technique compares profiles (i.e. the distribution of songbirds among distances) between 2 groups (years), testing 3 null hypotheses: 1) parallel distributions of songbirds among distances between years (i.e. no interaction between distance and year); 2) no differences in species richness or number of songbirds among distances (years pooled); and 3) no difference in species richness or number of songbirds between years (all distances pooled).

If a "riparian effect" were occurring, an increase in species richness or number of songbirds should be evident at sampling points near streams. I used complete-block-design ANOVA to test for differences in species richness and number of songbirds among distance from streams for each forest type (all sites) in each year (alpha = 0.05). I used site as the blocking variable because of potential high variability among sites. I tested residuals of each ANOVA for normality using the Kolmogorov D test (PROC UNIVARIATE, SAS 1982:575-583). I considered residuals normally distributed for $\underline{P} > 0.10$ and if homoscedasticity was not violated. Because I did not have replicates within each site and distance to test for an interaction between the 2 main effects (site and distance), I also used a model developed by Mountford (1982) for comparisons of sequential yearly counts of Common Bird Census (CBC) data, applying the model to my counts for distances. The Mountford model controls for a site effect by pairing counts within each site over all available distances. It then computes the relative effect (± 95% confidence intervals) of each distance based on ratios of within-site counts between each pair of distances. I converted the relative effect of each distance to mean number of birds (species or individuals) per distance by multiplying the relative effect produced by the model by the overall mean number of birds (all distances pooled). Data were checked for each set of bird parameters to ensure that they satisfied assumptions of the Mountford model.

The species composing the bird communities could vary between forest types and among distances. I compared the overall composition of bird communities observed between 1989 and 1990 and between floodplain and upland forests in 1990 with Jaccard's weighted ratio of similarity (SR) (Jongman et al. 1987:177):

$$SR_{ij} = \Sigma_k Y_{ki}Y_{kj}/(\Sigma_k Y_{ki}^2 + \Sigma_k Y_{kj}^2 - \Sigma_k Y_{ki}Y_{kj})$$

where Y_{ki} is the abundance of the kth species in year (or forest type) i, thereby comparing years (or forest types) i and j. This ratio of similarity encompasses both the presence of individual species and the number of individuals of each species, thereby resulting in a more precise measure of similarity between the 2 factors (years or forest types) being compared than an index relying only on presence of species. For each of the 6 most abundant species in each forest type, I tested for uniform distributions of individuals among distance from streams. I had adequate sample sizes for each abundant species in upland forests in 1989 ($\underline{n} \geq 30$ individuals) to use chi square goodness-of-fit tests. Because of small sample sizes for the most abundant species in each forest type in 1990 ($\underline{n} < 25$ individuals), I used Kolmogorov-Smirnov one-sample tests for discrete data (Zar 1984:53-55). I used a chi square test for homogeneity to compare the distribution of edge and interior individuals in floodplain and upland forests in 1990. For number of individuals of edge species and interior species, I tested for uniform distributions among distances from streams in each forest type with chi square goodness-of fit tests.

Habitat Characteristics

Vegetation sampling.--I sampled vegetation at 23 sites in July and August 1989 and at 5 sites in June 1990. I was unable to sample the twenty-fourth site in 1989 because it was logged before vegetation sampling began. Sampling was conducted along the transects at the same distances as bird censuses. At each distance, I recorded slope (flat, gentle, moderate, or steep) and aspect. Number of tree species, number of stems per tree species, and basal area were determined by the plotless method (Brower and Zar 1977) using a 2 m-factor prism. Using Strickler's (1959) method, I measured total percent canopy cover and percent conifer cover in the canopy with a densiometer. Heights of the canopy and subcanopy were determined by measuring the height of one dominant and one subdominant tree, respectively, with a Suunto clinometer. I counted number of stems of deciduous and of conifer shrubs per 50 m interval beginning from the stream using the line intercept method (Brower and Zar 1977). Each 50 m interval covered the diameter of a census plot, giving a total count of shrubs for each distance from the stream. I recorded presence or absence of 4 vertical strata (herbaceous: 0.0-0.3m; shrub: 0.3-3.0 m; subcanopy: 3.0-10.0 m; canopy: >10.0 m) at 10 m intervals from the stream, totalling 4 estimates per 50 m-interval. Frequencies of occurrence for each stratum and total number of strata were then averaged over the 4 estimates, giving a mean score per sampling point (distance from streams) ranging from 0-1 for each stratum, and 0-4 for total number of strata.

Statistical Analyses.--I used principal component analyses (PCA) to determine habitat gradients across all 28 sites from both years based on structural characteristics of the vegetation. PCA analyzes how variables either covary or vary independently, and then establishes gradients (component axes) based on these covarying variables. Each component that is established in the PCA is independent from every other component. Variables used in the PCA were number of tree species, basal area, total percent cover, percent conifer cover, canopy height, subcanopy height, ratio of subcanopy height to canopy height, number of conifer shrubs and of deciduous shrubs, presence of each vertical stratum, and total number of strata. I used the median value per site of each variable in the PCA. I retained only principal components (PC) that had eigen values >1.0 and explained >10% of total variance for interpretation and further analyses.

I tested for differences in means of each vegetation variable among the 6 distances from streams for each forest type in each year using Kruskal-Wallis tests (alpha = 0.05 for both tests).

Songbird-Riparian Habitat Associations

To determine the responses of birds to gradients in vegetation structure of sites, I used Spearman's rank correlation coefficients (r_s) to test for associations between each bird parameter and the principal component (PC) scores for all sites surveyed in each year (1989: $\underline{n}$ = 23 sites; 1990: $\underline{n}$ = 18 sites).

RESULTS

Distributions of Songbirds

I recorded 912 sightings of 55 songbird species at 24 study sites in 1989, and 580 sightings of 56 species at 18 sites in 1990. I also identified 42 individuals of 26 bird species at forest-stream interfaces (distance = 0 m to mid-stream), 9 species of which were not otherwise recorded within the forest: great blue heron, osprey, wood duck, common merganser, belted kingfisher, killdeer, spotted sandpiper, tree swallow, cliff swallow.

Patterns of songbird distributions (Fig. 2) and of community composition were similar in 1989 and 1990 for the 12 sites censused in both years. Repeated measures ANOVA detected no interaction between year and distance for either

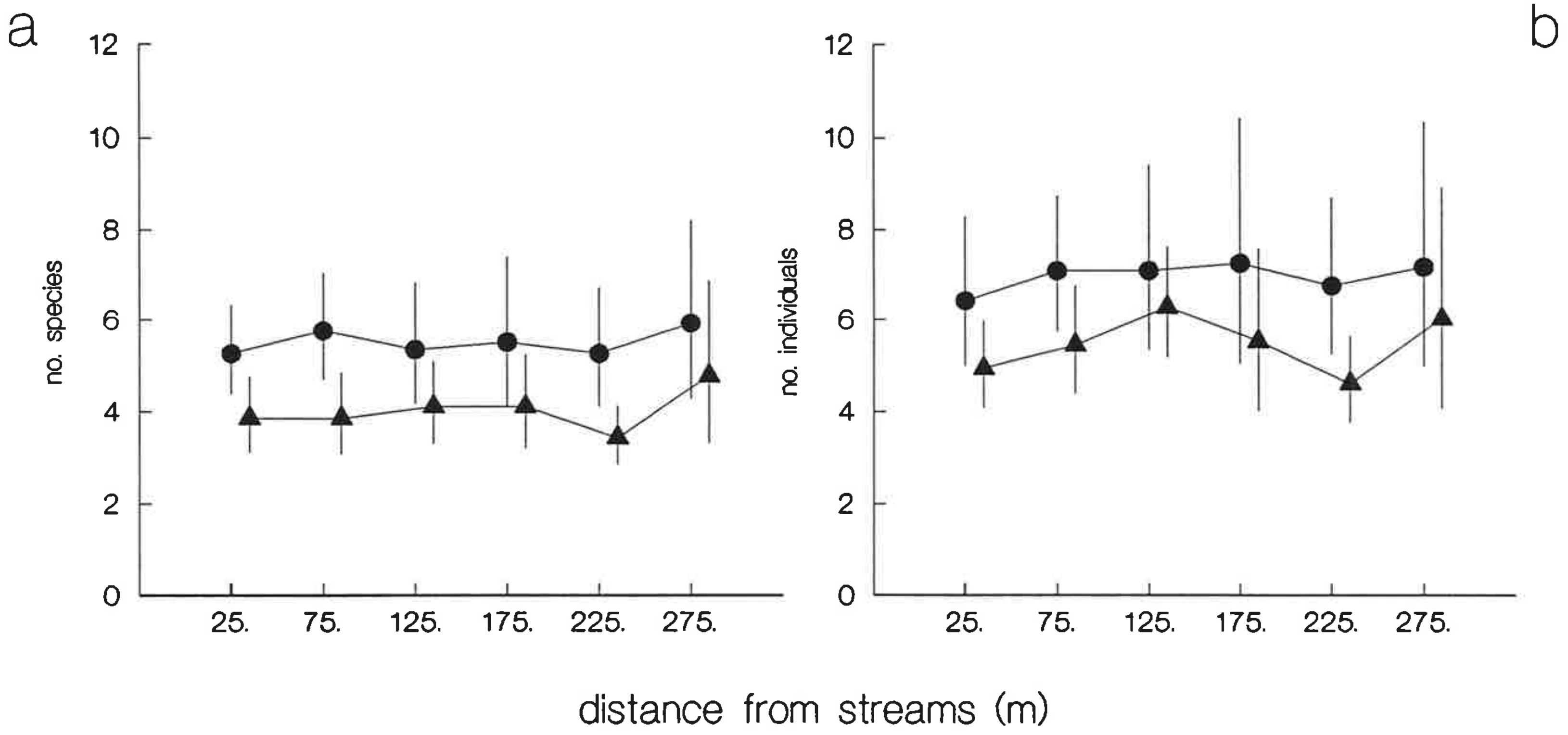

Figure 2. Distributions of songbirds relative to distance from streams for 12 riparian forest sites (Maine) censused both in 1989 and 1990. Circles denote 1989 results; triangles denote 1990 results. Points are means with 95% confidence intervals established by Mountford (1982) analysis: (a) number of species; (b) number of individuals.

species richness (F = 0.30, 5,18 df, P = 0.904) or number of individuals (F = 0.24, 5,18 df, P = 0.941), indicating parallel distributions in songbirds among distance from streams for these 2 years. A distance effect was not evident for either species richness (F = 0.86, 5,18 df, P = 0.529) or number of individuals (F = 0.66, 5,18 df, P = 0.657); however, I recorded more songbird species (F = 8.53, 1,22 df, P = 0.008) and individuals (F = 5.07, 1,22 df, P = 0.035) at these 12 sites in 1989 than in 1990. Jaccard's weighted ratio of similarity was 0.81 over all 12 sites, suggesting high similarity in community composition of birds between the 2 years. For these 12 sites, I recorded 8 species in 1989 not recorded in 1990, while all species recorded in 1990 were present in 1989. These results indicate significant consistency between years in the distributions of birds in these riparian forests, despite fewer birds being recorded in 1990.

Twenty-one of 24 sites I surveyed in 1989 were upland forests, while sites in 1990 were more evenly divided between floodplain (n = 8 sites) and upland forests (n = 10 sites). Preliminary analyses of data from 1989 indicated different patterns of songbird distributions relative to distance from streams between these 2 forest types. Hence, the proportion of sites of each forest type surveyed in each year substantially influenced results that were pooled over both forest types. In 1989, species richness and number of songbirds in floodplain sites decreased slightly with increasing distance from streams; however, because my sample size for floodplain sites in 1989 was so small (n = 3 sites), I did not include these sites in further analyses. Therefore, I have reported results from upland sites in 1989 (n = 21 sites) and 1990 (n = 10 sites), and floodplain sites in 1990 (n = 8 sites).

No trends relative to distance from streams were evident for species richness in upland forests, while, in floodplain forests, richness decreased with increasing distance from streams (Fig. 3a). In upland forests, species richness differed among distances in 1990 (F = 3.65, 5,45 df, P = 0.007), but not in 1989 (F = 0.811, 5,100 df, P = 0.545). Species richness also differed among sites in both years (1989: F = 2.85, 20,100 df, $P < 0.001$; 1990: F = 2.83, 9,45 df, P = 0.010). I used Mountford (1982) analysis, which controls for site effects by pairing within-site counts at each distance, for multiple comparisons between distances. Based on overlap of 95% confidence intervals, species richness was greater at 275 m than at 25 m from streams in upland forests in 1990, suggesting a slight increase in species richness with increasing distance from streams; however, with the larger sample size in 1989, no trend was evident (Fig. 3a). In contrast, species richness in floodplain forests (Fig. 3a) decreased significantly from 25 m to 125 m from streams, after which richness remained relatively even (F = 14.7, 5,35 df, $P < 0.001$). Species richness did not differ among sites in this forest type (F = 2.21, 7,35 df, P = 0.058).

Similar patterns relative to distance from streams occurred for number of individuals in each forest type as for species richness (Fig. 3b); however, this was not unexpected because I had few counts of multiple individuals for any species at the sampling points. Again, in upland forests, number of individuals differed

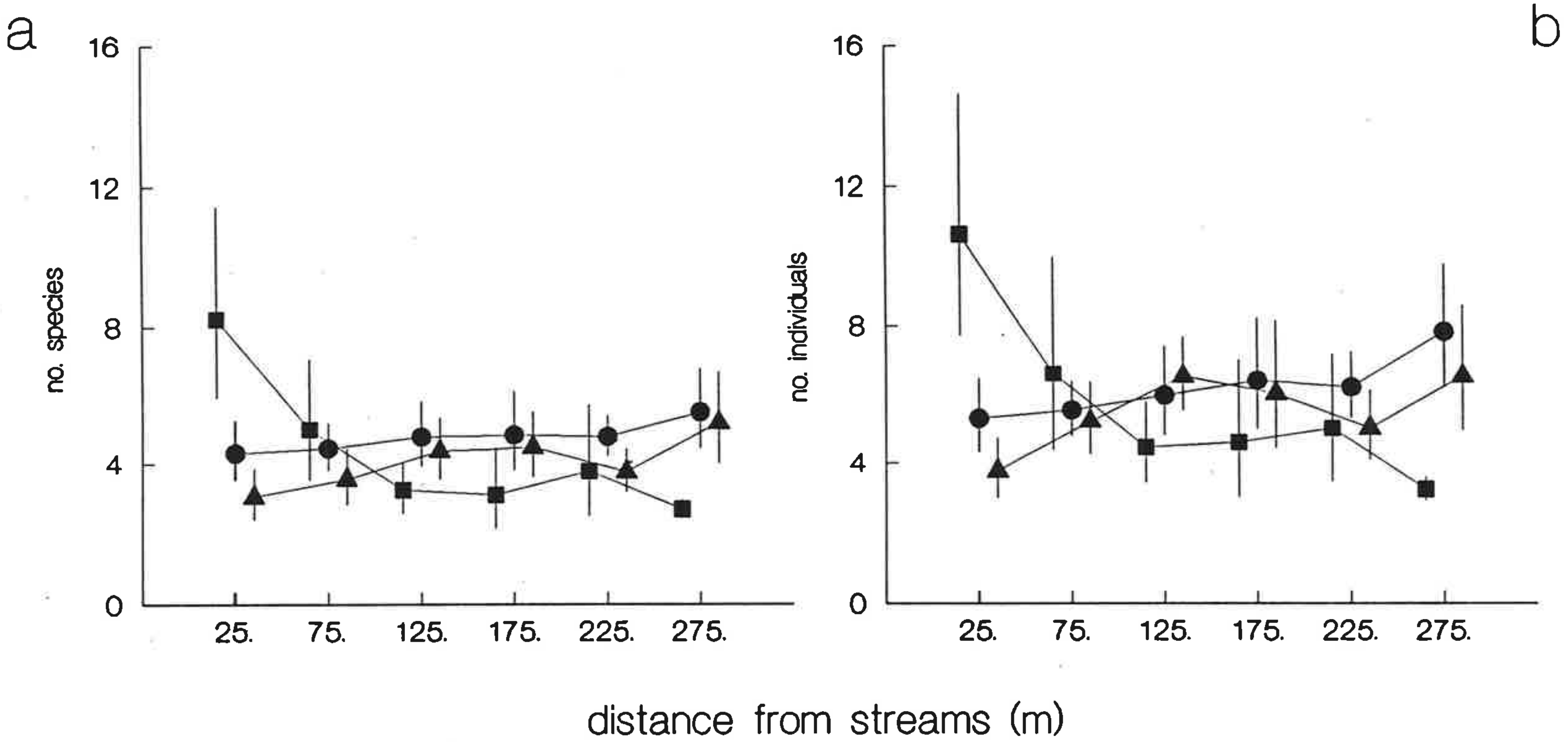

Figure 3. Distributions of songbirds relative to distance from streams in floodplain and upland forests, Maine, 1989-1990. Squares denote results from 8 floodplain sites in 1990; circles denote results from 21 upland sites in 1989; triangles denote results from 10 upland sites in 1990. Points are means with 95% confidence intervals established by Mountford (1982) analysis: (a) number of species; (b) number of individuals.

among distances in 1990 (*F* = 3.39,5,45 df, *P* = 0.011), but not in 1989 (*F* = 1.99, 5,100 df, *P* = 0.086), and no strong trends were evident in either year (Fig. 3b). Number of individuals differed among sites in 1989 (*F* = 2.38, 20,100 df, *P* = 0.003), but not in 1990 (*F* = 1.79, 9,45 df, *P* = 0.097). In floodplain forests, number of individuals decreased from 25 m to 125 m from streams, after which numbers remained relatively even (*F* = 11.8, 5,35 df, *P* < 0.001) (Fig. 3b). Number of individuals did not differ among sites (*F* = 1.97, 7,35 df, *P* = 0.087).

I also examined species recorded at each distance from streams for all sites combined that were not recorded at the previous distance (cumulative species richness), beginning 25 m from streams. By 225 m, I had recorded 98.1% and 92.7% of the species in upland forests in 1989 and 1990, respectively, and 97.7% of the species in floodplain forests in 1990 (Fig. 4; Appendix D)), indicating that any effects with distance occurred $\leq$ 225 m from streams.

The composition of the bird community also differed between the 2 forest types. Based on data from 1990 sites, 48% of the songbird community occurred within both forest types, indicating only moderate similarity between floodplain and upland forests. I observed 13 species only in upland forests, and 15 species only in floodplain forests. Twenty-eight species were common to both forest types (Appendix E). Furthermore, the most abundant species recorded in each forest type were quite different. Two abundant species common in both upland and floodplain forests, black-and-white warbler and black-capped chickadee, were uniformly distributed with distance from streams in each forest type (Table 2). In

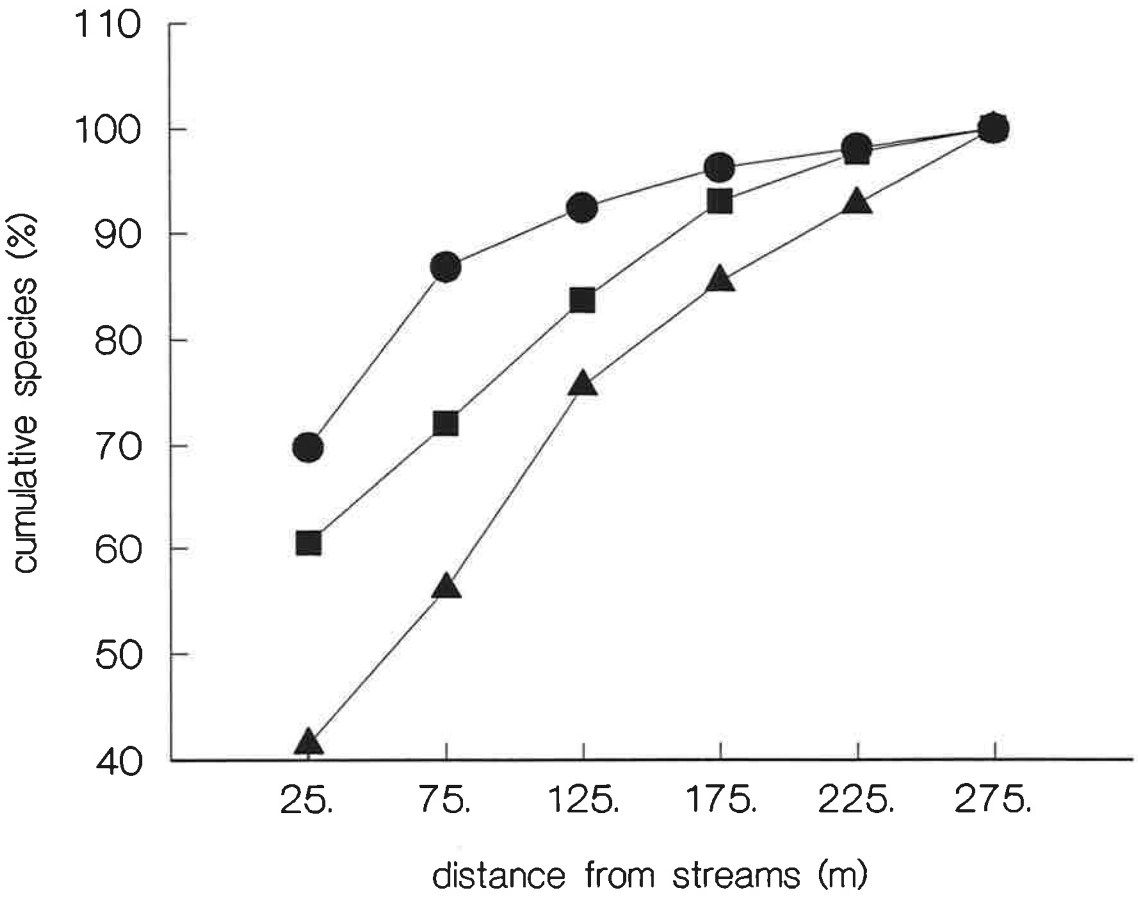

Figure 4. Cumulative curves for number of species relative to distance from streams in floodplain and upland forests, Maine, 1989-1990. Squares denote results from 8 floodplain sites in 1990; circles denote results from 21 upland sites in 1989; triangles denote results from 10 upland sites in 1990.

Table 2. Number of individuals of 6 most abundant bird species in upland[a] and floodplain[b] forests recorded in 25-m radius plots centered at 6 distances from streams, Maine, 1989-1990.

Forest type	Year	Bird species	Distance from streams (m)						$\underline{X}^2$ or $\underline{D}^c$
			25	75	125	175	225	275	
Upland	1989	black-and-white warbler	10	11	15	11	12	19	4.46
		ovenbird	7	7	11	18	12	20	11.96*[d]
		golden-crowned kinglet	7	12	13	9	7	7	4.02
		black-capped chickadee	6	7	7	13	8	11	4.31
		black-throated green warbler	1	6	8	12	8	12	10.83
		red-eyed vireo	7	4	4	4	7	7	2.45
	1990	black-and-white warbler	4	11	11	9	5	8	4.0
		black-capped chickadee	4	8	4	7	6	5	1.7
		ovenbird	3	4	2	3	3	7	3.3
		golden-crowned kinglet	3	4	3	5	1	3	2.3

Table 2. Continued.

Forest type	Year	Bird species	Distance from streams (m)						X^2 or D^c
			25	75	125	175	225	275	
Floodplain	1990	black-throated blue warbler	2	2	3	2	2	4	1.5
		black-throated green warbler	2	2	2	3	4	6	3.7
		black-capped chickadee	6	5	2	2	7	2	3.0
		black-and-white warbler	4	3	3	4	4	5	1.5
		least flycatcher	5	5	2	5	1	1	4.3
		common yellowthroat	6	5	3	1	0	1	6.0^{*}
		Northern waterthrush	8	6	1	0	0	0	9.0^{***}
		veery	6	2	0	1	5	1	3.5

[a] 1989: $n = 21$ sites/distance, 1990: $n = 10$ sites/distance

[b] $n = 8$ sites/distance

[c] For upland forests in 1989, I had adequate sample sizes ($n \geq 30$ individuals/species) to use chi square goodness-of-fit tests (X^2). For upland and floodplain forests in 1990, I used Kolmogorov-Smirnov one-sample tests (D) for discrete data because of small sample sizes ($n < 25$ individuals for all but 2 species). Both techniques tested for uniform distributions among distances.

[d] * $P < 0.05$, *** $P < 0.001$

upland forests, abundant species also included ovenbird, golden-crowned kinglet, black-throated green warbler, red-eyed vireo, and black-throated blue warbler; all species except ovenbird were uniformly distributed with distance from streams (Table 2). Ovenbird increased as distance increased in 1989, but was independent of distance in 1990. Additional species abundant in floodplain forests included least flycatcher and veery, both uniformly distributed with distance, and common yellowthroat and northern waterthrush, both of which decreased as distance increased (Table 2).

Differences in community composition were further exemplified in the distribution of individuals of edge and interior species. In 1990, I recorded 218 individuals for the 14 species that I classified as either edge or interior species. Interior species (96 of 123 individuals, 78.0%) dominated in upland forests, while edge species (63 of 95 individuals, 66.3%) dominated in floodplain forests ($\underline{X}^2$ = 43.5, 1 df, $\underline{P} < 0.001$). Within floodplain forests, individuals of edge species decreased with increasing distance from streams ($\underline{X}^2$ = 23.8, 5 df, $\underline{P} < 0.001$), with Mountford analysis indicating more edge individuals <225 m from streams (Fig. 5a). In contrast, individuals of edge species in upland forests in both 1989 ($\underline{X}^2$ = 9.94, 5 df, $0.05 < \underline{P} < 0.10$) and 1990 ($\underline{X}^2$ = 5.22, 5 df, $\underline{P} > 0.25$) were distributed uniformly among distance from streams (Fig. 5a). Similarly, individuals of interior species in floodplain forests ($\underline{X}^2$ = 6.62, 5 df, $\underline{P} > 0.20$) and in upland forests in both 1989 ($\underline{X}^2$ = 4.82, 5 df, $\underline{P} > 0.25$) and 1990 ($\underline{X}^2$ = 4.62, 5 df, $\underline{P} > 0.25$) were distributed uniformly among distance from streams (Fig. 5b).

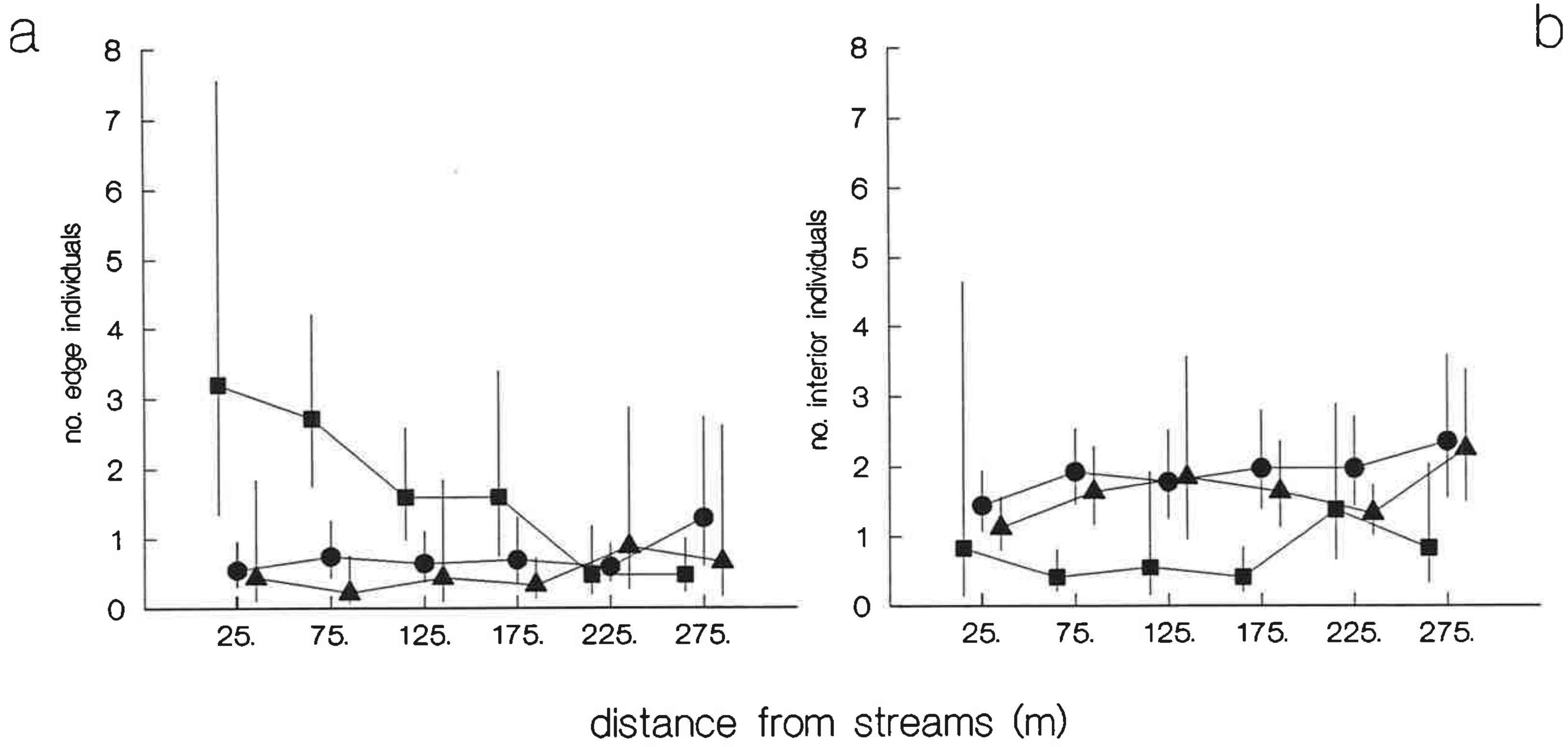

Figure 5. Distributions of edge and interior songbirds relative to distance from streams in floodplain and upland forests, Maine, 1989-1990. Squares denote results from 8 floodplain sites in 1990; circles denote results from 21 upland sites in 1989; triangles denote results from 10 upland sites in 1990. Points are means with 95% confidence intervals established by Mountford (1982) analysis: (a) number of individuals classified as edge species; (b) number of individuals classified as interior species.

Habitat Characteristics

Substantial differences were present in the vegetation structure of the 28 sites, as evidenced by PCA. The first 3 principal components each had eigen values >2.0, while all other components had eigen values $\leq$1.0. I retained the first 3 components, which explained 71.7% of the total variance in vegetation structure (Table 3), for interpretation. Sites at the positive end of the first component (PC1) contained many tree species, dominated by conifers; trees were tall, with large basal area, and relatively closed canopy (Table 3). Sites at the negative end of PC1 contained fewer tree species and smaller trees, but had well-developed vertical structure (e.g. many strata), being particularly dominated by herbaceous and deciduous shrub layers (Table 3). Thus, PC1 generally reflected a gradient in forest types from coniferous forests to deciduous forests. This component, explaining nearly half the variation in vegetation structure, was clearly the dominant gradient in my sites. Sites at the positive end of the second component (PC2) had taller canopy and subcanopy strata, with sites at the negative end of PC2 having shorter upper strata, and a more distinct subcanopy stratum (Table 3). Thus, PC2 reflected a gradient in vertical structure for the upper strata of the sites. Sites at the positive end of the third component (PC3) had a well-developed canopy that provided substantial canopy closure (percent canopy cover), and, conversely, sites at the negative end of PC3 had a more open canopy (Table 3). Thus, PC3 reflected a gradient in the horizontal structure for the upper strata of the sites. PC1 clearly separated floodplain and upland sites

Table 3. Component loadings for the first 3 principal components (PC) of vegetation characteristics for 28 sites in riparian forests, Maine, 1989-1990.

	PC1[a]	PC2[b]	PC3[c]
Eigen Value	6.0	2.2	1.9
% Variance	42.5	15.8	13.4
Vegetation variable			
No. tree species	0.85	-0.01	-0.05
Basal area (m^2/ha)	0.85	0.31	0.05
Canopy cover (%)			
total	0.63	-0.20	0.55
conifer	0.80	0.04	-0.12
Height (m)			
canopy	0.71	0.45	0.26
subcanopy	-0.11	0.90	0.16
Subcanopy ht.:canopy ht.	-0.81	0.36	-0.17
No. shrubs			
conifer	0.62	-0.14	0.38
deciduous	-0.59	-0.14	0.38
Presence of vertical strata			
herbaceous	-0.73	0.24	-0.18
shrub	-0.62	-0.35	0.14
subcanopy	0.23	-0.76	0.42
canopy	0.26	0.36	0.74

Table 3. Continued.

	PC1	PC2	PC3
Mean no. strata	-0.70	0.08	0.50

[a] PC1 represents a gradient in forest type (coniferous to deciduous).
[b] PC2 represents a gradient in the height of the upper (i.e. canopy, subcanopy) strata.
[c] PC3 represents a gradient in canopy closure.

(Fig. 6a). Aside from the forest-type gradient, sites were not distinctly grouped along PC2 or PC3, but composed a continuum along each of these 2 components (Fig. 6b-c). In general, upland sites had more tree species, with greater basal area and canopy cover, a more dominant conifer component, and fewer deciduous shrubs than floodplain sites (Table 4). Only height of subcanopy was similar between the 2 forest types.

In upland forests, structure of the vegetation did not vary with increasing distance from streams in either 1989 (Kruskal-Wallis, $\underline{P} \geq 0.36$ for each vegetation variable) or 1990 ($\underline{P} \geq 0.14$ for each vegetation variable). However, in floodplain forests, number of tree species ($\underline{X}^2 = 20.1$, 5 df, $\underline{P} = 0.001$), percent conifer cover ($\underline{X}^2 = 14.4$, 5 df, $\underline{P} = 0.013$), and number of conifer shrubs ($\underline{X}^2 = 15.8$, 5 df, $\underline{P} = 0.008$) each increased as distance increased (Fig. 7). This deciduous-coniferous gradient with distance in floodplain forests is similar to the gradient detected in sites along PC1.

Songbird-Riparian Habitat Associations

Distributions of songbirds in riparian forests may be confounded by variations in vegetation of these forests. Correlations between bird parameters and principal component axes indicated that birds responded to gradients in the vegetation structure of sites (Table 5). The negative association of number of edge individuals with forest type (PC1) suggested that edge species responded positively (i.e. with increased abundance) to vegetation structure of floodplain sites. The positive association of number of interior individuals with forest type

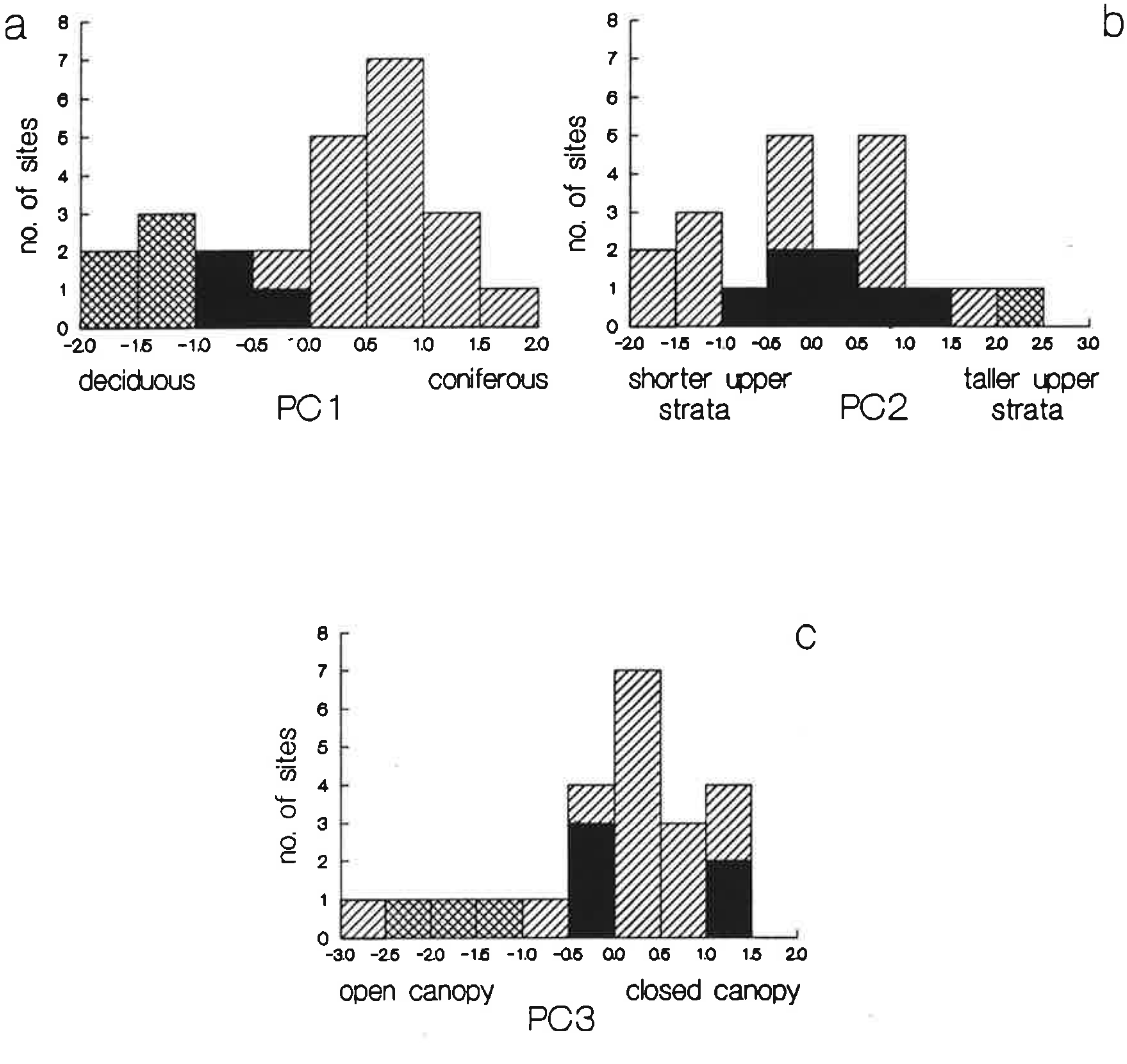

Figure 6. Plots of frequencies of 28 sites classified as upland (n = 20) or floodplain (n = 8) against scores of the first 3 principal component (PC) axes, Maine, 1989-1990. Angled lines denote upland sites; cross-hatchings denote floodplain sites; solid bars denote overlap between forest types. PCs establish gradients in the vegetation structure of these sites: (a) PC1 represents a gradient in forest type (coniferous to deciduous); (b) PC2 represents a gradient in the vertical structure of the upper (i.e. canopy, subcanopy) strata; (c) PC3 represents a gradient in the horizontal structure (i.e. percent cover) of the canopy.

Table 4. Mean and standard error for vegetation variables recorded in floodplain and upland sites, Maine, 1989-1990.

	Floodplain sites[a]		Upland sites[b]			
	1990		1989		1990	
Vegetation Variable	x	S.E.	x	S.E.	x	S.E.
No. tree species	3.62	0.22	5.62	0.15	5.33	0.11
Basal area (m^2/ha)	24.17	1.26	32.10	0.87	31.62	0.76
Canopy cover (%)						
total	33.04	2.23	59.63	1.34	59.03	0.94
conifer	6.20	1.58	27.32	2.30	24.40	1.53
Height (m)						
canopy	17.93	0.43	22.69	0.40	21.69	0.28
subcanopy	9.25	0.29	8.87	0.21	8.61	0.15
Subcanopy ht.:canopy ht.	0.53	0.02	0.40	0.01	0.40	0.01
No. shrubs						
conifer	7.02	1.71	22.25	2.16	22.78	1.82
deciduous	36.10	3.39	15.43	2.71	24.61	2.98
Presence of vertical strata						
herbaceous	0.85	0.03	0.45	0.04	0.53	0.03
shrub	0.46	0.04	0.27	0.03	0.32	0.02
subcanopy	0.45	0.03	0.60	0.03	0.59	0.02
canopy	0.72	0.04	0.90	0.02	0.85	0.02
Mean no. strata	2.46	0.07	2.21	0.05	2.27	0.04

[a] n = 8 sites x 6 distances = 48

[b] 1990: n = 10 sites x 6 distances = 60; 1989: n = 20 sites x 6 distances = 120

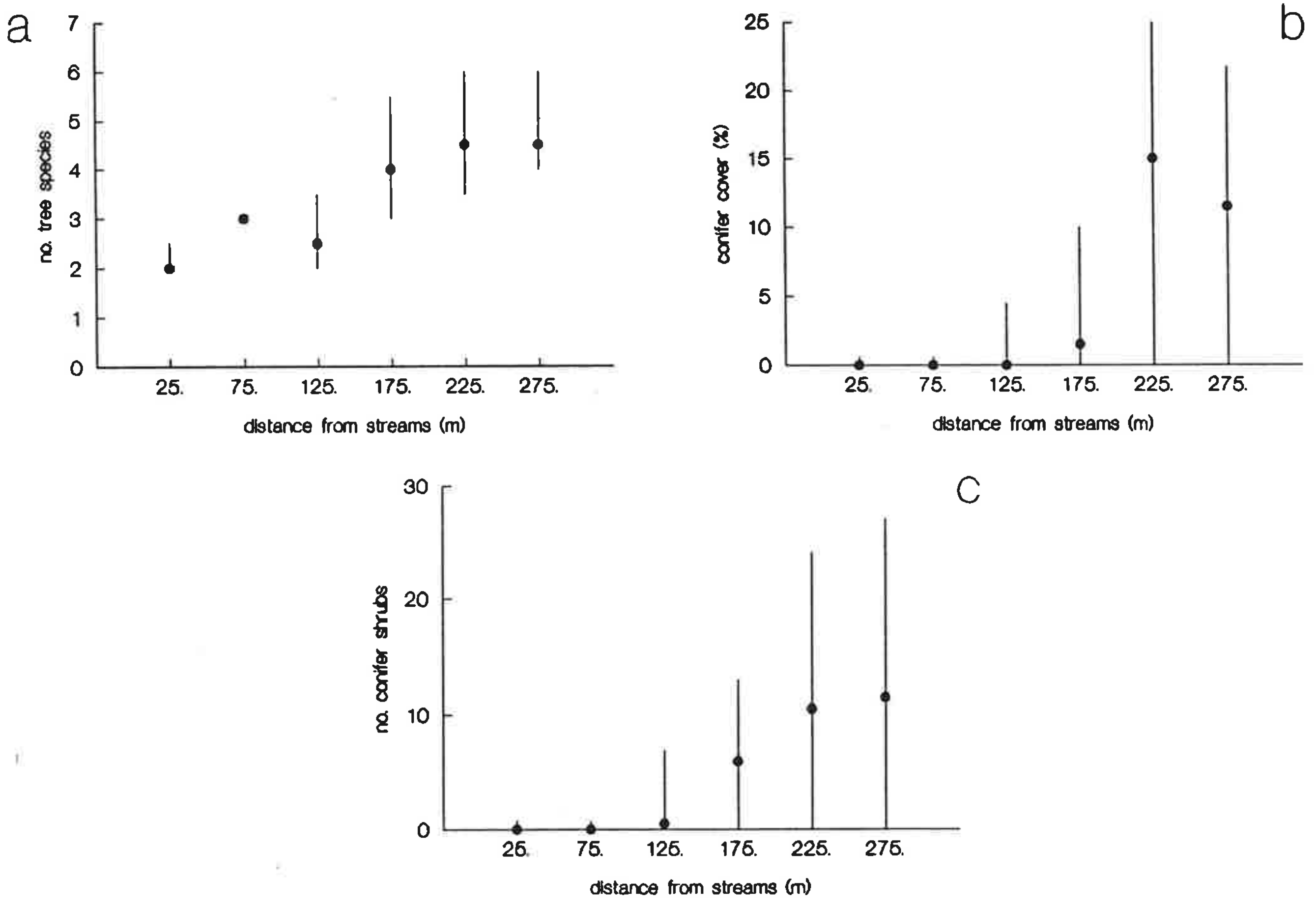

Figure 7. Medians ± 25th and 75th quantiles for 3 vegetation variables in floodplain forests (n = 8 sites) relative to distance from streams, Maine, 1990: (a) number of tree species; (b) percent conifer cover; (c) number of conifer shrubs.

Table 5. Correlations (r_s) between songbird parameters and the first 3 principal components[a] (PC) in riparian forests, Maine, 1989-1990.

Bird variable	Year	PC1	PC2	PC3
No. species	1989	-0.20	-0.28	-0.48^{**b}
	1990	-0.14	-0.16	-0.54^{*}
No. individuals	1989	-0.08	-0.06	-0.38^{*}
	1990	0.06	-0.24	-0.34
No. edge individuals	1989	-0.45^{*}	-0.20	-0.08
	1990	-0.79^{***}	-0.01	-0.42^{*}
No. interior individuals	1989	0.38^{*}	-0.15	-0.24
	1990	0.57^{**}	0.01	0.28

[a] 1989: $n = 23$ sites/correlation; 1990: $n = 18$ sites/correlation

[b] $P < 0.05$, $P < 0.01$, $P < 0.001$ (Spearman's rank correlation)

(PC1) suggested that interior species responded positively to vegetation structure of upland forests. The negative association of species richness and numbers of songbirds with canopy closure (PC3) suggested that openings in the canopy strata, regardless of forest type, attracted a greater richness and abundance of songbirds than closed-canopy sites.

Furthermore, within floodplain sites, decreases in richness and numbers of all songbirds (Fig. 3) and edge species (Fig. 5a) from 25 m to 125 m from streams corresponded to increases in conifers and number of tree species (Fig. 7) with increasing distance. Apparently, the open, shrubby vegetation nearer streams at floodplain sites attracted a greater richness and abundance of songbirds, primarily edge species, which decreased with the transition to coniferous upland forest further from streams. At distances $\geq$ 175 m from streams, conifers dominated at most floodplain sites and no further decreases in songbird richness or abundance were evident.

DISCUSSION

Riparian habitats are generally characterized as transitional zones with vegetative features distinct from adjacent aquatic and upland habitats (Brinson et al. 1981, Ohmart and Anderson 1986). These zones provide greater structural diversity in vegetation by maximizing edges in the transition from aquatic to upland habitats (Thomas et al. 1979, Small and Johnson 1986). Although this description applies to riparian habitats in the western U.S., many of the sites I studied lacked this transition in vegetation. Upland sites, which comprised 21 of

29 sites, contained abrupt interfaces between streams and forest habitats, with little change in vegetation as distance from streams increased. Presence of streams did not seem to influence structure of the vegetation at upland sites, and thus these forests were relatively indistinguishable from non-riparian forests of central Maine. In contrast, floodplain sites provided a transitional habitat characterized by more open, shrubby vegetation, and more standing water than upland sites or adjacent non-riparian forests. The floodplain sites I studied more closely fit the typical description of riparian habitats located in the western U.S. (Thomas et al. 1979, Johnson and Lowe 1985, Ohmart and Anderson 1986). Many of the deciduous floodplain sites in my study did not extend >200 m from streams. By this distance, floodplain sites often had an established conifer component. Despite the broad-scale differences in vegetation and hydrology between floodplain and upland forests, all sites in my study constituted general gradients in forest structure, as evidenced by their distributions along the principal component axes.

Odum (1971:159) defined "edge effect" as an increase in number of wildlife species, density of individuals, or diversity of the community within an ecotone. In my study, a similar "riparian effect" was evidenced by increased species richness and abundance of all songbirds and edge species in floodplain sites, but not in upland sites. Previous studies conducted in the western U.S. also documented riparian effects for songbirds in riparian habitats that form distinct ecotones (review by Brinson et al. 1981; Szaro and Jakle 1985, Knopf 1985, Hunter et al.

1987, Finch 1989). In these western riparian zones, presence of water, increased plant biomass and habitat complexity, and numerous edges compared to adjacent upland habitats contributed to the influx of species (Thomas et al. 1979, Brinson et al. 1981, Swift et al. 1984, Finch 1989).

In my study, patterns of songbird distributions in upland forests seemed more closely related to structure of the vegetation than to distance from streams. Without a distinct ecotone, or even a narrow transitional edge at the forest-stream interface, lack of a riparian effect in upland forests was not surprising. Although species richness and total individuals were not related to distance from streams, songbirds did seem to respond to the relative amount of cover provided by the canopy of sites. Structural heterogeneity of vegetation often increases in canopy openings, attracting species not usually present in closed-canopy forests (Swift et al. 1984, Johnson and Brown 1990). My results agreed with those of other studies of songbirds conducted along abrupt edges in Maine. Species richness and density of songbirds were similar between the forest interior and abrupt edges created by estuarine rivers at 2 sites in southern Maine (Small 1986) and by clearcutting in north-central Maine (Elliot 1987). Elliot (1987) also attributed the lack of an edge effect to the lack of a shrubby ecotone along these abrupt edges.

In floodplain forests in my study, the effect of a riparian ecotone was most evident 25 m from streams. Mean species richness and total individuals at this distance were >2 times greater in floodplain forests than in upland forests. In a study in Iowa, Stauffer and Best (1980) reported that number of species was similar between floodplain and upland woodland sites, although densities of breeding birds were substantially higher in the floodplain sites. Swift et al. (1984) documented that, in forested wetlands, the most poorly drained sites had the most abundant and diverse bird populations. The greater water content of floodplain forests may increase species richness and abundance of songbirds either directly, by providing more available water, or indirectly, by affecting vegetation structure and composition. These direct and indirect effects together may produce a greater variety of niches and a greater amount of food, such as seeds or insects (Gaines 1974, Brown et al. 1978, Swift et al. 1984). I did not have data from my sites for these parameters to further examine this hypothesis.

All songbird species detected in my study occur in other habitats in central Maine. Only 2 of 9 species detected only at forest-stream interfaces were passerine species (tree swallow, cliff swallow). These 2 swallow species, as well as the 2 songbird species (common yellowthroat, northern waterthrush) that had significantly more individuals near streams in floodplain forests, occur in non-riparian habitats in central Maine (DeGraaf and Rudis 1987). The remaining 7 species observed only at forest-stream interfaces may be more dependent upon riparian edges, and need further consideration in future studies of and regulations for riparian zones in Maine.

The songbird community in upland forests in both 1989 and 1990 comprised primarily species that occupy interior forests in northeastern U.S. (Titterington 1977, Johnson and Brown 1990, Elliot 1987). Edge species, although not uncommon, were generally less abundant than interior species. Transitional edges that characterize riparian ecotones in more arid regions of the U.S. were not present in upland forests to attract bird species specific to riparian habitats (Thomas et al. 1979, Brinson et al. 1981). In my study, most abundant species in upland forests were independent of distance from streams, responding to the general cover type (conifer, mixed-wood, deciduous) of forests rather than to any direct influence of streams.

In contrast, the songbird community in floodplain forests of my study comprised primarily edge and generalist species, with interior species contributing relatively little to overall composition or abundance. Four of 6 most abundant species in floodplain forests were the same as those in forested wetlands of Massachusetts (Swift et al. 1984), whereas only 2 of 6 most abundant species were common between floodplain and upland forests in my study in central Maine. In west-central U.S., greater species diversity of songbirds in riparian zones compared to upland habitat were attributed to greater structural complexity of riparian vegetation (review by Brinson et al. 1981, Ohmart and Anderson 1986). However, the bird species that inhabited these zones were often generalists or species with cosmopolitan distributions (Knopf 1986, Finch 1989). This pattern was evident for birds inhabiting floodplain forests in my study. In general, the open, shrubby vegetation provided by floodplain sites attracted edge species. As conifers and number of tree species increased with the transition to upland forest at floodplain sites, edge species declined.

SUMMARY AND CONCLUSIONS

I investigated distributions of songbirds among 6 distances from streams in floodplain and upland forests, with the null hypothesis that species richness and number of songbirds would not vary with increasing distance from streams (i.e. no riparian effect). My results suggested that the distributions of songbirds in riparian forests of central Maine were significantly influenced by patterns of forest vegetation adjacent to streams.

Floodplain sites were dominated by open, shrubby deciduous vegetation ≤ 175 m from streams (Fig. 6, Table 4). Floodplain forests occurred as transitional habitats between stream and upland habitats; conifers and number of tree species increased with increasing distance from streams (Fig. 7), and at ≥ 175 m from streams, most floodplain sites had an established conifer component. Changes in songbird abundance and richness with increasing distance from streams in floodplain forests paralleled these changes in vegetation structure. Species richness (Fig. 3a) and abundance of all songbirds (Fig. 3b) and edge species (Fig. 5a) decreased from 25 m to 125 m from streams; > 125 m from streams, no further decline was evident.

Upland riparian sites were dominated by large, coniferous trees, with a closed canopy (Fig. 6, Table 4). Upland forests formed an abrupt edge with streams, and had little change in vegetation with increasing distance from streams. I detected no trend in species richness (Fig. 3a) or abundance of all songbirds (Fig. 3b), edge species (Fig. 5a), or interior species (Fig. 5b) relative to distance from streams in upland forests. Interior species dominated the songbird community in upland forests, while edge species dominated the songbird community in floodplain forests.

Lack of a transitional ecotone in upland riparian sites, combined with the lack of a riparian effect in species richness or abundance of songbirds, exemplifies the difficulty in extrapolating results from studies in western riparian habitats to conditions in Maine. In upland riparian forests of central Maine, the number, richness, and species composition of songbirds do not appear to change with distance from streams. This is in direct contrast to western studies, where the presence of riparian habitats significantly increases regional songbird diversity and abundance (Brinson et al. 1981, Szaro and Jakle 1985, Knopf 1986, Hunter et al. 1987, Finch 1989). My study, however, neither adequately addressed reproductive or foraging success of songbirds, nor life requirements of non-passerine bird species (e.g. waterfowl, wading birds) using riparian forests of central Maine.

MANAGEMENT IMPLICATIONS

My results indicated a minimum 200 m buffer strip is necessary to retain all songbird species using undisturbed riparian forests of central Maine, if maintaining the entire bird community is the goal of regulations for riparian forests. More than 90% of songbird species in both 1989 and 1990 were present $\leq$225 m from streams (Fig. 4). In addition, harvesting operations in a riparian buffer strip that create openings in the canopy may increase species richness and abundance of songbirds, but the increase may be primarily generalist species, possibly to the detriment of some interior forest species (Stauffer and Best 1980, Brinson et al. 1981, Howard and Allen 1988, this study).

Regulations requiring 200 m buffer strips along inland shores might restrict timber harvesting on substantial amounts of land. Beyond minimum buffer widths to maintain aquatic ecosystems, managing upland forests for songbirds may be better incorporated into plans that manage the surrounding landscape (Knopf 1986, Knopf et al. 1988, Howard and Allen 1988). If regulations for buffer strips are not wide enough (e.g. ≤200 m) to maintain many bird species, particularly interior species, then the surrounding landscape should be managed to provide sufficient forest for these species. Managing on a landscape scale may be difficult in Maine because 96% of the timberland is privately owned, with 47% owned by the forest industry (Brooks et al. 1986).

Floodplain forests could be managed as distinct vegetative zones. Because of their similarity to wetland forests (Brown et al. 1978, Swift et al. 1984), regulations protecting wetlands should include stipulations for floodplain forests.

Future research should focus on distributions of songbirds (and other terrestrial wildlife) in buffer strips of different widths to determine to what degree species will decrease their territory sizes or increase habitat overlap among species (Finch 1989). I conducted my study in relatively undisturbed forests, with adequate habitat for many interior forest species. Further research is needed to determine at what widths and total acreages individual songbird species will no longer inhabit a riparian buffer strip. In addition, future studies should address the potential of riparian forests in central Maine to provide life requirements for non-passerine bird species that may be more dependent on this type of habitat, and to provide travel corridors for mammals.

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CHAPTER III

EFFECTS OF STREAM NOISE ON EFFICIENCY OF BREEDING BIRD CENSUSES

During breeding bird censuses, observers generally rely on singing male birds for identifying and counting songbird species. Studies have examined the influences of hearing ability of observers (Cyr 1981, Faanes and Bystrak 1981, Ramsey and Scott 1981), attenuation of bird songs by forest vegetation (Richards 1981, Waide and Narins 1988), and bird abundance (Scott and Ramsey 1981, Bart and Schultz 1984) on song recognition and counts of birds. Although researchers have acknowledged that background noise may also influence counts, most reports are only anecdotal (Cyr 1981, Dawson 1981, Emlen and Dejong 1981, Waide and Narins 1988). Background noises caused by a stream may decrease the probability of detecting birds during censuses near the stream. My study of songbird use of riparian forests in Maine (Chapter II) included sites with different levels of background noise caused by streams. The objective of this chapter was to evaluate the effect of noise from streams on the efficiency of my bird censuses.

STUDY AREAS

I classified streams as loud or quiet based on the relative amount of noise created by the water in the streams. Loud streams were $\leq$15 m wide, with numerous large rocks in the streambed over which the water flowed. The banks and surrounding terrain of loud streams were moderately steep ($\geq$20° rise). The sound of water rushing within narrow streambeds and over rocks created the "loud" background noise. Quiet streams were $\geq$20 m wide, with slow moving water in a sandy or silty streambed. The surrounding terrain was generally flat or with a slight rise ($\leq$10°). A wide streambed and no rocks resulted in little or no background noise.

I selected 4 sites next to streams classified as loud and 4 sites next to quiet streams. Sites were located in central Maine (see Fig. 1: quiet sites were no. 5, 20, 21, and 24; loud sites were no. 6, 15, 28, 29). Sites adjacent to loud streams were deciduous or coniferous upland forests. Sites adjacent to quiet streams were either deciduous floodplain forests or mixed-wood upland forests. Coniferous upland sites were dominated by eastern hemlock (Tsuga canadensis), balsam fir (Abies balsamea), and/or red spruce (Picea rubens). Deciduous upland sites were dominated by red maple (Acer rubrum), sugar maple (Acer saccharum), and American beech (Fagus grandifolia). Mixed-wood sites were dominated by a combination of these conifer and deciduous species. All upland sites had $\geq$60% canopy cover and $\leq$30% shrub cover. Floodplain sites were dominated by red maple or silver maple (Acer saccharinum), with $\leq$30% canopy cover and $\geq$50% shrub cover.

METHODS

I chose 8 bird species to represent 2 song volumes and 4 song patterns: loud-continuous (Troglodytes troglodytes); loud-staccato (Seiurus aurocapillus); loud-repetitive (Vireo olivaceus); loud-single note (call note of Catharus fuscenscens); quiet-continuous (Dendroica fusca); quiet-staccato (Regulus satrapa); quiet-repetitive (Sitta canadensis); and quiet-single note (Sphyrapicus varius). Songs of these 8 species were recorded onto a cassette tape from an album of Peterson's Field Guide to Bird Songs of Eastern and Central North America (Houghton Mifflin Co., Boston, Mass., 1983) in random order at intervals from 15-30 seconds between songs. Prior to the test, the volume setting of the tape recorder was adjusted in the field at a quiet stream to resemble the natural volume of a loud-staccato song (Seiurus aurocapillus) at 25 m. The site at which I adjusted the volume was a mixed-wood upland site with 25% shrub cover and 50% canopy cover. The same volume setting was used at all sites. I conducted the test during mid-day in June 1990. This time frame avoided confusing the taped songs with actual bird songs and ensured that water levels in the streams were comparable to those during my study.

I established 3 transects perpendicular to the stream at 50 m intervals at each site. I tested my census efficiency along each transect at 3 listening points located 25 m, 75 m, and 125 m from the stream. At distances >125 m even the loudest streams were barely audible. While I stood at each listening point, an assistant played the tape recorder for 3 minutes 25 m away in the direction of the stream. Thus, when I stood 25 m from the stream, the assistant was at the stream edge; when I was at 75 m, the assistant was at 50 m; and when I was at 125 m, the assistant was at 100 m. I conducted my field censuses within 25 m-radius plots, so this design tested a "worst-case scenario". The assistant recorded all songs played during each listening interval, and I recorded all songs that I heard during that time. The test was conducted once at each site.

Based on initial results, I selected the loudest stream for more intensive testing (see Fig. 1, site no. 29). I established listening points at 25 m, 75 m, and 125 m along 6 transects set perpendicular to the stream at 25 m intervals. The same test design was used as in the extensive (multiple-site) test, except that the tape was played for 10 minutes for each listening period. I concentrated most effort in this test at 25 m from the stream. Listening points at 25 m were each tested 3 times; listening points at 75 m and 125 m were each tested once.

For each test, total number of songs played versus heard were summed for each song type at each distance. Statistical tests were based on the intensive test because small sample sizes prevented valid testing of the extensive test. I used one-tailed Fisher exact tests (Zar 1984:390-393) to compare the percent of songs not heard close to the stream (25 m) with the percent not heard further from the stream (>25 m) (alpha = 0.05). These comparisons were conducted for each song type and all song types combined. I used a Poisson distribution (Snedecor and Cochran 1980:130-133) to estimate probabilities that I did not hear birds in my field censuses (alpha = 0.05). I calculated probabilities using proportions of birds not heard (no. songs not heard/no. songs played) determined from the intensive test, with sample sizes obtained during my 1989 and 1990 breeding bird censuses (Chapter II).

RESULTS

I heard all songs at 75 m and 125 m in both the extensive and intensive tests; therefore, for each test, I combined results for these distances. At quiet streams, I heard all songs that were played. However, at loud streams, I did not hear 2 of 46 songs (4.3%) at 25 m, including 1 of 7 loud-single note and 1 of 7 soft-single note in the extensive test (Table 6). In the intensive test, I did not hear 1.4% of songs ($\underline{n}$ = 372) at 25 m, including 6.0% ($\underline{n}$ = 50) of soft-continuous songs and 4.4% ($\underline{n}$ = 46) of soft-stacatto songs (Table 6). I did not detect differences between the percent of songs not heard at 25 m from the stream and at 75 m and 125 m for soft-continuous songs ($\underline{P}$ = 0.268) or for soft-stacatto songs ($\underline{P}$ = 0.312). For all song types combined, I heard fewer songs near the stream than farther from it; however, this trend was not significant ($\underline{P}$ = 0.070). I heard a greater percent of loud songs than soft songs at 25 m from the loudest stream ($\underline{P}$ = 0.026).

My sample sizes of bird abundance at 25 m from the stream that were obtained from breeding bird censuses in 1989-1990 (Chapter II) ranged from 1-4 individuals per species and 1-13 individuals of all species for each site. Based on estimates from the Poisson distribution, I had a <0.03 probability of not hearing >1 bird with a soft song at 25 m from a loud stream during my censuses (Table 7). For all song types, I had a <0.04 probability of not hearing >1 bird at 25 m from a loud stream during my censuses (Table 7).

Table 6. Proportion of songs not heard (no. not heard/no. played) at 25 m and >25 m from streams during tests of effects of stream noise on census efficiency, Maine, 1990.

Song type		Extensive test[a]				Intensive test[b]	
		Distance (m) from quiet streams		Distance (m) from loud streams		Distance (m) from loud stream	
volume	pattern	25	>25	25	>25	25	>25
Loud	Continuous	0/6	0/11	0/5	0/12	0/44	0/28
	Stacatto	0/4	0/11	0/5	0/12	0/50	0/33
	Repetitive	0/7	0/10	0/5	0/11	0/46	0/30
	Single note	0/6	0/11	1/7	0/13	0/52	0/33
Soft	Continuous	0/7	0/11	0/5	0/11	3/50	0/27
	Stacatto	0/5	0/11	0/7	0/9	2/46	0/36
	Repetitive	0/6	0/13	0/4	0/13	0/42	0/34
	Single note	0/5	0/11	1/7	0/11	0/42	0/39
All song types		0/46	0/99	2/45	0/92	5/372	0/260

[a] Extensive test included sites at 4 quiet streams and 4 loud streams.

[b] Intensive test included only one site at the loudest stream.

Table 7. Estimated probabilities (P) of not hearing songbirds with different song types at 25 m from a loud stream during breeding bird censuses[a]. Sample sizes (n) reflect those obtained during breeding bird censuses in riparian forests, Maine, 1989-1990; proportions of songs not heard were calculated for 8 song types from an auditory test[b].

No. of individuals (n)	Song type					
	Soft-continuous		Soft-stacatto		All song types	
	$\leq X^c$	P	$\leq X^c$	P	$\leq X^c$	P
1	1	0.998	0	0.957	0	0.987
2	1	0.993	1	0.996	0	0.974
3	1	0.986	1	0.992	0	0.960
4	1	0.975	1	0.986	1	0.999
5	1	0.963	1	0.980	1	0.998
6	2	0.994	1	0.971	1	0.997
7	2	0.991	1	0.962	1	0.996
8	2	0.987	1	0.952	1	0.995
9	2	0.982	2	0.992	1	0.993
10	2	0.977	2	0.990	1	0.992
11	2	0.970	2	0.987	1	0.990
12	2	0.963	2	0.984	1	0.988
13	2	0.955	2	0.980	1	0.986
14	3	0.989	2	0.976	1	0.984
15	3	0.986	3	0.995	1	0.982

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[a] Probabilities (P) of not hearing birds were estimated using a Poisson distribution.

[b] For 6 song types, proportions of songs not heard (no. songs not heard/no. songs played) were 0.00. Proportions of songs not heard were 0.060 (3/50) for soft-continuous songs, 0.043 (2/46) for soft-stacatto songs, and 0.013 (5/372) for all song types combined.

[c] X = number of birds possibly not heard during a census at 25 m from a loud stream.

DISCUSSION

The low numbers of songs not being heard suggests that background noises caused by loud, rushing streams probably had no appreciable effect on my census efficiency of breeding birds within 25-m plots in riparian forests in Maine. Dawson (1981) also reported no significant effect of environmental noise on counts of Grey Warblers (Gerygone igata) in New Zealand. Of my 28 sites, I classified only 4 as loud streams. Therefore, in my study, I can be confident that few, if any, birds were not heard during my actual breeding bird censuses because of river noise.

I heard loud songs more than soft songs, indicating that stream noises have a greater effect on an observer detecting soft songs than loud songs. Both soft songs that I did not hear were high frequency songs, whereas the other 6 bird songs tested were low frequency songs. Humans have greater difficulty hearing high frequency songs than low frequency songs, regardless of their different lengths and complexities (Waide and Narins 1988).

While these results suggest that the effects of river noises on breeding bird censuses in riparian forests in Maine were insignificant, potential biases from background noise should not be ignored under other censusing conditions. Background noises with different frequencies (e.g. rivers, highways, loud machinery, livestock concentrations) may selectively effect the probability of detecting different bird songs (Emlen and DeJong 1981). In addition, hearing abilities of observers may confound effects of background noise (Cyr 1981).

Finally, the method of censusing (e.g. variable circular plots, fixed-radius plots, strip transects) and size of census plots could in part determine the influence of background noises during a bird census (Edwards et al. 1981, Waide and Narins 1988).

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APPENDICES

APPENDIX A. BIRD SPECIES, THEIR SCIENTIFIC NAMES, AND NUMBER OF SIGHTINGS IN 1989 AND 1990 BIRD CENSUSES.

		No. sightings	
Common name	Scientific name	1989	1990
Black-and-white warbler	Mniotilta varia	87	71
Ovenbird	Seiurus aurocapillus	80	28
Golden-crowned kinglet	Regulus satrapa	58	22
Black-capped chickadee	Parus articapillus	57	58
Black-throated green warbler	Dendroica virens	49	19
Red-eyed vireo	Vireo olivaceus	39	21
Veery	Catharus fuscenscens	33	25
American robin	Turdus migratorius	31	18
Hermit thrush	Catharus guttatus	31	14
Least flycatcher	Empidonax minimum	30	28
Blackburnian warbler	Dendroica fusca	26	6
Northern parula	Parula americana	25	18
Black-throated blue warbler	Dendroica caerulescens	24	15
Yellow-bellied sapsucker	Sphyrapicus varius	21	12
Magnolia warbler	Dendroica magnolia	21	5
Canada warbler	Wilsonia canadensis	20	8
Common yellowthroat	Geothlypis trichas	19	17
Blue jay	Cyanocitta cristata	19	5
American redstart	Setophaga ruticilla	19	4
White-throated sparrow	Zonotrichia albicollis	17	4
Winter wren	Troglodytes troglodytes	17	5
Red-breasted nuthatch	Sitta canadensis	16	16
Cedar waxwing	Bombycilla cedrorum	13	3
Yellow-rumped warbler	Dendroica coronata	13	6
Solitary vireo	Vireo solitarius	12	6
Eastern wood-pewee	Contopus virens	11	6
Common grackle	Quiscalus quiscula	10	2
Downy woodpecker	Picoides pubescens	9	7
Yellow warbler	Dendroica petechia	8	8
Northern waterthrush	Seiurus noveboracensis	8	17

APPENDIX A. CONTINUED.

Common name	Scientific name	No. sightings 1989	1990
White-breasted nuthatch	Sitta carolinensis	7	1
Rose-breasted grosbeak	Pheucticus ludoviciansus	7	4
Great crested flycatcher	Myiarchus crinitus	7	7
Chestnut-sided warbler	Dendroica pensylvanica	7	10
Swainson's thrush	Catharus ustulatus	7	4
Nashville warbler	Vermivora ruficapilla	6	2
American crow	Corvus brachyrhynchos	6	1
Bay-breasted warbler	Dendroica castanea	5	1
Eastern kingbird	Tyrannus tyrannus	-[a]	5
Brown creeper	Certhia americana	4	8
Scarlet tanager	Piranga olivacea	4	1
Northern flicker	Colaptes auratus	4	1
Swamp sparrow	Melospiza georgiana	-	4
Alder flycatcher	Empidonax alnorum	3	3
Wilson's warbler	Wilsonia pusilla	3	2
Pileated woodpecker	Dryocopus pileatus	3	-
Pine warbler	Dendroica pinus	3	-
Ruby-crowned kinglet	Regulus calendula	-	3
Yellow-bellied flycatcher	Empidonax flaviventris	2	3
Purple finch	Carpodacus purpureus	2	-
Northern oriole	Icterus galbula	2	1
Hairy woodpecker	Picoides villosus	2	1
Warbling vireo	Vireo gilvus	-	2
Yellow-throated vireo	Vireo flavifrons	1	-
Tennessee warbler	Vermivora peregrina	1	3
Song sparrow	Melospiza melodia	1	1
Red-winged blackbird	Agelanius phoeniceus	1	3
Evening grosbeak	Coccothraustes vespertinus	1	-
Eastern phoebe	Sayornis phoebe	1	-
Black-billed cuckoo	Coccyzus erythropthalmus	-	1
Gray catbird	Dumetella carolinensis	-	1
Wood thrush	Hylocichla mustelina	-	1

APPENDIX A. CONTINUED.

Common name	Scientific name	No. sightings	
		1989	1990
Riparian edge only[b]			
Belted kingfisher	Ceryle alcyon	-	P[c]
Cliff swallow	Petrochelidon pyrrhonata	-	P
Common merganser	Mergus merganser	P	-
Great-blue heron	Ardea herodias	P	-
Killdeer	Charadrius vociferus	P	P
Osprey	Pandion haliaetus	P	P
Spotted sandpiper	Actitis macularia	P	P
Tree swallow	Tachycineta bicolor	P	-
Wood duck	Aix sponsa	P	-

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[a] bird species not recorded in that year.

[b] bird species observed only between the forest-stream interface and middle of stream.

[c] bird species present (recorded) in that year (counts of individuals were not recorded).

APPENDIX B. PRELIMINARY STUDY OF SONGBIRD DISTRIBUTIONS IN RIPARIAN FORESTS, MAINE, 1987.

The original protocol for this study was an intensive survey of 3 riparian sites in Maine. Preliminary data collected during summer, 1987, indicated potential difficulties with this study design. I, therefore, changed the protocol for the remainder of the study to an extensive survey of riparian sites in Maine (Chapter II). The objectives of this appendix are to outline the preliminary study, present qualitative results, and discuss the problems which led to the change in study design.

I wish to gratefully acknowledge Sumner Roberts, who collected the data for the 1987 preliminary study.

Study Areas.--Two streams and one pond (>10 ha) located in central Maine were chosen for the study. The Blackman Stream (Bradley, Penobscot County) and Chemo Pond sites (Eddington, Penobscot County) were both located on the Penobscot Experimental Forest. The Oyster River site (Warren, Knox County) was located on private land.

All sites were moderately to well-drained, with mixed-wood or coniferous overstories. Dominant tree species were red spruce (Picea rubens), balsam fir (Abies balsamia), and hemlock (Tsuga canadensis). Each site, however, comprised substantial habitat patchiness. For Blackman Stream, patches were generally created by regeneration from spruce-budworm kills. For the other 2 sites, heterogeneity in the vegetation was often a result of localized differences in drainage patterns, creating a substantial American beech (Fagus grandifolia)-sugar maple (Acer saccharum) patch at the Chemo Pond site, and numerous alder (Alnus sp.) or white cedar (Thuja occidentalis) patches at the Oyster River site.

Methods.-- Transects were established along the east-west magnetic directions at 50 m intervals (generally perpendicular to the shore). The Blackman Stream site had 12 transects, the Chemo Pond site had 11 transects, and the Oyster River site had 8 transects. Each transect was then marked at 25 m intervals for sampling stations.

Five surveys of breeding birds at each site were conducted on non-stormy mornings in June, 1987, beginning within one-half hour of sunrise. Starting points were varied to avoid time-of-day bias. Observers stopped for 3 minutes at 50 m intervals along every other transect (e.g. 100 m apart), and recorded all birds heard or seen, estimating direction and distance from the sampling point.

I did not statistically analyze these data; however, I examined trends in number of species and percent incidence of the most abundant bird species. Although vegetation data was also collected in 1987, I did not analyze or evaluate these data.

Qualitative Results. The number of breeding bird species relative to distance from water varied among sites (Fig. B1). At the Blackman Stream site, more species were breeding near the stream, and fewer as distance from water increased (Fig. B1a). At the Oyster River site, exactly the opposite trend

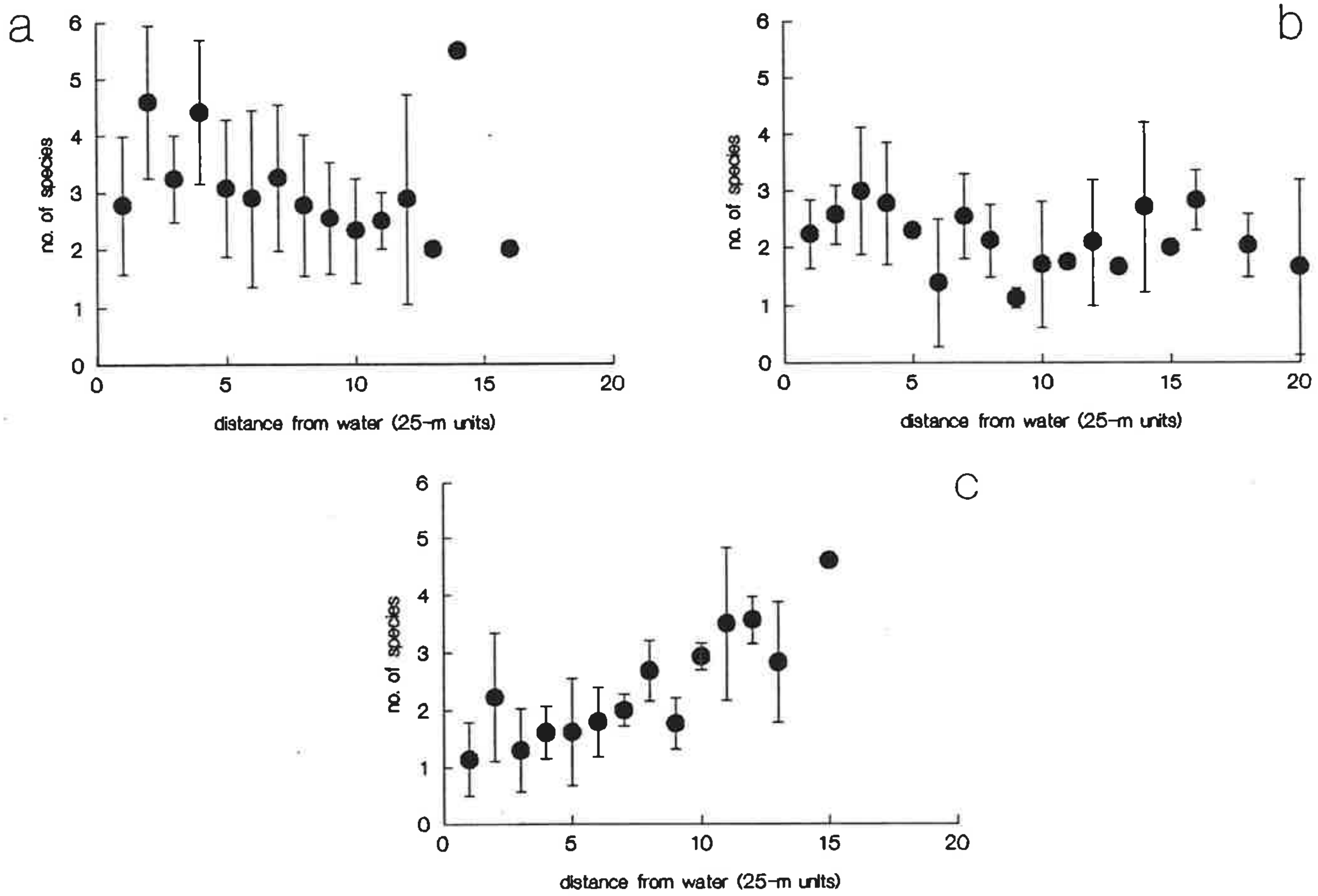

Figure B1. Mean (+ 1 S.D.) number of species relative to distance from water for 3 riparian forest sites, Maine, 1987: (a) Blackman Stream, Penobscot County; (b) Chemo Pond, Penobscot County; (c) Oyster River, Knox County.

occurred, with few species breeding near the stream, and more as distance from water increased (Fig. B1c). No trend was evident for the Chemo Pond site (Fig. B1b).

Distributions of the abundant species within a site also varied greatly. For example, at the Blackman Stream site, Northern parulas (Parula americana) were universally present closer to the stream (within 100 m), decreasing as distance from water increased (Fig. B2a). In contrast, ovenbirds (Seirus motacilla) at Blackman Stream were not present within the first 25 m of the stream, but their incidence elsewhere at the site was ubiquitous (Fig. B2b).

Conclusions. The site and species specificity documented in this preliminary study made generalizing how Maine's riparian forests influence bird communities difficult. The type of water body (e.g. stream or pond) may differentially influence local vegetation, and ultimately the distributions of birds. Hence, using 2 streams and 1 pond may confound patterns detected among the sites. Sampling only 3 sites may reduce the applicability of the results to other parts of Maine or the Northeast. In addition, the habitat patchiness within the sites may have substantially influenced the distributions of birds. Sites with more homogeneous vegetation may have better controlled for some influences of vegetation. However, during summer, 1988, I could not locate large, homogeneous, relatively undisturbed riparian sites that would have been adequate for an intensive study.

These difficulties prompted me to design an extensive survey of riparian

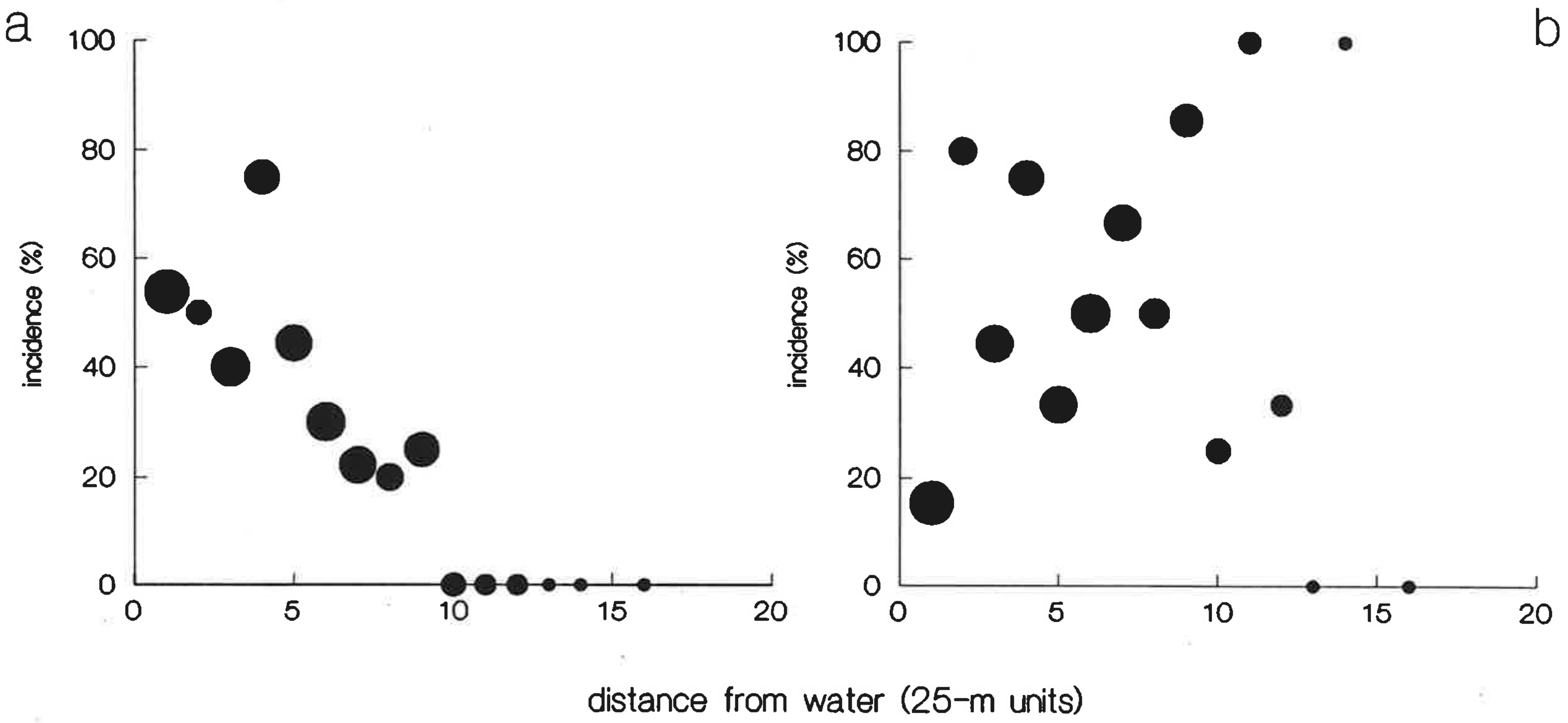

Figure B2. Percent incidence relative to distance from water for 2 of the most abundant species recorded during breeding bird surveys at Blackman Stream, Penobscot County, Maine, 1987. Circle size reflects number of points sampled at a given distance: (a) Northern parula; (b) ovenbird.

forests in Maine. I limited the study to forests adjacent to streams, and incorporated numerous small sites that were relatively undisturbed and homogenous in their vegetation. An extensive survey design could provide more insight into the generality of patterns for species or bird communities breeding in riparian forests of Maine.

APPENDIX C. EVALUATION OF SPECIES RICHNESS AND COMPOSITION OF BREEDING BIRD COMMUNITIES BETWEEN FIRST AND SECOND CENSUSES IN 1989 IN RIPARIAN FORESTS, MAINE.

Objective.--To test for consistency between multiple censuses within a breeding bird season, I compared number of species and community composition between the first and second breeding bird censuses of 14 sites censused twice in 1989 (Chapter II).

Statistical Analyses.--I tested for differences in number of species per census plot of each site (n = 18 plots/site for each census) between the first and second censuses using Mann-Whitney U tests (alpha = 0.05). I also used Mann-Whitney U tests to test for differences in number of species at each distance between the first and second censuses (n = 14 sites/distance for each census). Over all sites, I tested for differences in number of species between the two censuses with analysis of covariance (ANACOVA) (Wilkinson 1987) on rank-transformed data (Conover and Iman 1981), using distance as the covariate. No interaction occurred between census number (1 or 2) and distance ($P = 0.105$). Also, for each site, I qualitatively checked for differences in the species and number of individuals of each species identified between the first and second censuses for each site.

Results.--Only 3 of 14 sites censused twice in 1989 had significantly different numbers of species per census plot between the first and second censuses (Table C1). At 1 site, I recorded more species per census plot in the first than in

Table C1. Means and standard errors of number of songbird species per census plot (*n* = 18 plots/site) for study sites with 2 breeding bird censuses in 1989 in riparian forests, Maine.

Site Code[b]	Census 1[a]		Census 2[a]		Mann-Whitney U	*P*
	x	S.E.	*x*	S.E.		
BS1	2.22	0.26	2.61	0.20	121.0	0.165
BS2	2.00	0.28	1.83	0.30	165.0	0.922
EBS1	1.72	0.31	2.28	0.34	123.5	0.214
EBS2	1.44	0.26	1.50	0.27	159.0	0.925
MBU1	3.44	0.54	1.89	0.31	228.0	0.033
MBU2	2.17	0.42	1.50	0.20	171.0	0.767
OS	1.83	0.33	1.50	0.25	181.0	0.534
PUS	1.72	0.28	2.39	0.30	105.5	0.068
RS	1.06	0.26	1.83	0.28	101.5	0.048
SER	2.22	0.28	1.89	0.34	190.5	0.355
SOS1	1.67	0.30	1.72	0.33	156.0	0.845
SOS2	2.17	0.36	1.94	0.30	176.5	0.639
WBU	1.28	0.25	3.29	0.39	52.5	<0.001
WS1	1.17	0.18	1.44	0.23	131.0	0.300

[a] "Census 1" refers to the first census conducted at a site during the 1989 breeding bird season; "Census 2" refers to the second census conducted at a site during the 1989 breeding bird season.

[b] Table 3 (Chapter II) identifies site code and location for each site.

the second census. At 2 sites, the opposite trend occurred. Number of species was not significantly different between the first and second census at any distance from streams (Table C2). Overall, the ranks of number of species between the 2 censuses were not significantly different when adjusting for distance ($\underline{F} = 2.826$, 1,165 df, $\underline{P} = 0.095$). For 46 of 75 (61.3%) instances when I recorded a species for the second census but not for the first census at a site, I only observed one individual of that species. In only 4 (5.3%) instances did I record >4 (but ≤8) individuals of a species during the second census, but no individuals in the first census at a site. For all sites combined, I did not record any species in the second censuses that I had not observed in the first censuses.

Conclusion.--These results suggest significant consistency between the 2 censuses in the 1989 breeding bird season. Hence, using only one census per site should not substantially affect the overall results of this study.

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Table C2. Means and standard errors of number of songbird species per distance from streams of 14 study sites with 2 breeding bird censuses in 1989 in riparian forests, Maine.

Distance from streams (m)	Census 1[a]		Census 2[a]		Mann-Whitney U	P
	x	S.E.	x	S.E.		
25	1.38	0.26	1.81	0.25	64.5	0.118
75	1.26	0.14	1.64	0.24	70.0	0.192
125	1.86	0.30	1.93	0.22	84.0	0.516
175	2.02	0.28	2.19	0.22	82.0	0.457
225	2.07	0.37	2.00	0.19	86.5	0.593
275	2.60	0.16	2.41	0.24	114.0	0.554

[a] "Census 1" refers to the first census conducted at a site during the 1989 breeding bird season; "Census 2" refers to the second census conducted at a site during the 1989 breeding bird season.

APPENDIX D. DISTRIBUTIONS OF SONGBIRD SPECIES WITH DISTANCE FROM STREAMS IN RIPARIAN FORESTS, MAINE, 1989-1990.

Table D1. Number of individuals of each songbird species per distance from streams for 21 sites, combined, in upland forests, Maine, 1989.

	Distance from streams (m)					
Bird Species	25	75	125	175	225	275
Black-and-white warbler	10	11	15	11	12	19
Ovenbird	7	7	11	18	12	20
Golden-crowned kinglet	7	12	13	9	7	7
Black-capped chickadee	6	7	7	13	8	11
Black-throated green warbler	1	6	8	12	8	12
Red-eyed vireo	7	4	4	4	7	7
Hermit thrush	5	5	3	3	5	7
Blackburnian warbler	4	3	1	3	6	9
American robin	4	1	3	3	6	7
Black-throated blue warbler	2	2	2	4	5	9
Northern parula	2	7	4	2	5	3
Veery	4	4	3	3	6	2
Magnolia warbler	2	3	4	4	5	3
Yellow-bellied sapsucker	4	3	5	2	4	1
Canada warbler	3	2	2	1	1	8
Least flycatcher	0	4	4	3	4	2
Blue jay	4	3	0	1	0	8
American redstart	2	3	3	2	5	1
Winter wren	2	1	7	2	2	1
Red-breasted nuthatch	0	1	3	4	5	2
White-throated sparrow	1	4	2	2	1	4
Common yellowthroat	1	1	4	3	0	4
Yellow-rumped warbler	1	2	3	3	0	3
Solitary vireo	3	3	1	2	1	1
Swainson's thrush	5	0	0	1	1	0
Common grackle	4	1	0	0	2	0
White-breasted nuthatch	1	0	0	4	0	1
American crow	0	0	1	1	3	1

Table D1. Continued.

	Distance from streams (m)					
Bird Species	25	75	125	175	225	275
Cedar waxwing	2	0	1	2	0	1
Red-breasted grosbeak	1	1	0	2	0	2
Downy woodpecker	2	0	1	2	0	0
Eastern wood-pewee	1	3	0	0	0	1
Bay-breasted warbler	0	3	0	1	0	1
Chestnut-sided warbler	0	1	0	2	1	1
Nashville warbler	0	0	0	1	3	1
Brown creeper	0	2	2	0	0	0
Great-crested flycatcher	2	1	0	0	0	1
Scarlet tanager	0	1	1	1	1	0
Alder flycatcher	2	0	1	0	0	0
Pine warbler	1	0	1	1	0	0
Pileated woodpecker	0	1	1	0	1	0
Common flicker	0	1	0	1	0	1
Yellow warbler	1	1	0	0	0	0
Hairy woodpecker	0	0	0	1	1	0
Purple finch	2	0	0	0	0	0
Northern waterthrush	2	0	0	0	0	0
Eastern phoebe	1	0	0	0	0	0
Wilson's warbler	1	0	0	0	0	0
Yellow-throated vireo	0	1	0	0	0	0
Yellow-bellied flycatcher	0	0	1	0	0	0
Evening grosbeak	0	0	1	0	0	0
Northern oriole	0	0	0	0	1	0
Song sparrow	0	0	0	0	0	1
Total no. species	37	36	33	37	30	35
Cumulative no. species	37	46	49	51	52	53

Table D2. Number of individuals of each songbird species per distance from streams for 3 sites, combined, in floodplain forests, Maine, 1989.

	Distance from streams (m)					
Bird Species	25	75	125	175	225	275
Least flycatcher	3	4	6	0	0	0
Veery	3	2	1	0	4	1
Black-and-white warbler	4	0	1	2	2	0
American robin	1	0	3	2	1	0
Cedar waxwing	0	0	2	3	2	0
Northern waterthrush	4	2	0	0	0	0
Red-eyed vireo	1	1	2	1	0	1
Eastern wood-pewee	1	1	1	3	0	0
Yellow warbler	3	2	1	0	0	0
Common yellowthroat	1	3	2	0	0	0
Black-capped chickadee	2	1	0	2	0	0
Ovenbird	0	0	0	2	0	3
Downy woodpecker	0	1	1	2	0	0
Hermit thrush	0	0	0	0	0	3
Common grackle	1	1	1	0	0	0
White-throated sparrow	1	1	0	1	0	0
Blue jay	0	1	0	0	0	2
Great-crested flycatcher	0	0	1	1	1	0
Golden-crowned kinglet	0	0	0	1	1	1
American redstart	0	0	0	0	1	2
Canada warbler	0	0	0	0	0	3
Chestnut-sided warbler	1	0	1	0	0	0
Yellow-bellied sapsucker	1	0	0	1	0	0
Wilson's warbler	0	2	0	0	0	0
Northern parula	0	0	0	0	1	1
Black-throated green warbler	0	0	0	0	1	1
Winter wren	0	0	0	0	1	1
Red-winged blackbird	1	0	0	0	0	0
Northern oriole	1	0	0	0	0	0
White-breasted nuthatch	0	1	0	0	0	0
Norhtern flicker	0	0	1	0	0	0
Red-breasted nuthatch	0	0	0	1	0	0

Table D2. Continued.

	Distance from streams (m)					
Bird Species	25	75	125	175	225	275
Solitary vireo	0	0	0	0	1	0
Yellow-rumped warbler	0	0	0	0	1	0
Yellow-bellied flycatcher	0	0	0	0	1	0
Tennessee warbler	0	0	0	0	0	1
Nashville warbler	0	0	0	0	0	1
Rose-breasted grosbeak	0	0	0	0	0	1
Total no. species	16	14	14	13	13	14
Cumulative no. species	16	20	23	26	33	38

Table D3. Number of individuals of each songbird species per distance from streams for 10 sites, combined, in upland forests, Maine, 1990.

	Distance from streams (m)					
Bird Species	25	75	125	175	225	275
Black-and-white warbler	4	11	11	9	5	8
Black-capped chickadee	4	8	4	7	6	5
Ovenbird	3	4	2	3	3	7
Golden-crowned kinglet	3	4	3	5	1	3
Black-throated green warbler	2	2	2	3	4	6
Black-throated blue warbler	2	2	3	2	2	4
Hermit thrush	0	4	2	1	3	4
Northern parula	1	1	4	2	4	1
Red-breasted nuthatch	0	2	6	1	1	3
Red-eyed vireo	1	1	3	3	1	3
Veery	2	1	3	1	0	3
Least flycatcher	0	0	1	0	3	5
American robin	3	2	0	2	2	0
Yellow-bellied sapsucker	3	0	0	1	1	1
Blackburnian warbler	0	0	2	1	2	1
Blue jay	1	0	2	0	1	1
Winter wren	1	0	1	2	0	1
Yellow-rumped warbler	0	1	0	2	0	2
Magnolia warbler	0	1	0	1	2	0
Swainson's thrush	0	0	2	2	0	0
Eastern wood-pewee	0	1	2	0	0	0
Solitary vireo	0	0	1	2	0	0
Chestnut-sided warbler	0	0	1	1	1	0
American redstart	0	0	1	0	1	1
Canada warbler	0	0	0	2	0	1
Northern waterthrush	2	0	0	0	0	0
Great-crested flycatcher	1	1	0	0	0	0
Brown creeper	1	0	0	1	0	0
Gray catbird	1	0	0	0	0	0
Ruby-crowned kinglet	0	1	0	0	0	0
Yellow-bellied flycatcher	0	1	0	0	0	0
Warbling vireo	0	0	1	0	0	0

Table D3. Continued.

	Distance from streams (m)					
Bird Species	25	75	125	175	225	275
American crow	0	0	1	0	0	0
Scarlet tanager	0	0	0	1	0	0
Song sparrow	0	0	0	1	0	0
Rose-breasted grosbeak	0	0	0	0	1	0
Hairy woodpecker	0	0	0	0	1	0
Common yellowthroat	0	0	0	0	1	0
Downy woodpecker	0	0	0	0	0	1
Wood thrush	0	0	0	0	0	1
Nashville warbler	0	0	0	0	0	1
Total no. species	17	18	22	24	21	22
Cumulative no. species	17	24	32	35	38	41

Table D4. Number of individuals of each songbird species per distance from streams for 8 sites, combined, in floodplain forests, Maine, 1990.

	Distance from streams (m)					
Bird Species	25	75	125	175	225	275
Black-capped chickadee	6	5	2	2	7	2
Black-and-white warbler	4	3	3	4	4	5
Least flycatcher	5	5	2	5	1	1
Common yellowthroat	6	5	3	1	0	1
Veery	6	2	0	1	5	1
Northern waterthrush	8	6	1	0	0	0
American robin	4	2	2	0	0	1
Red-eyed vireo	0	2	2	1	1	3
Yellow warbler	1	2	2	2	1	0
Chestnut-sided warbler	4	2	1	0	0	0
Yellow-bellied sapsucker	3	2	1	0	0	0
Brown creeper	1	0	1	2	1	1
Downy woodpecker	1	2	2	0	1	0
Ovenbird	0	0	1	1	2	2
Great-crested flycatcher	3	1	0	0	1	0
Eastern kingbird	3	0	1	1	0	0
Canada warbler	0	1	0	0	2	2
Northern parula	0	0	1	1	1	2
Swamp sparrow	4	0	0	0	0	0
White-throated sparrow	0	1	0	2	1	0
Red-breasted nuthatch	0	0	0	0	3	0
Alder flycatcher	3	0	0	0	0	0
Red-winged blackbird	3	0	0	0	0	0
Rose-breasted grosbeak	2	1	0	0	0	0
Cedar waxwing	2	0	0	0	1	0
Eastern wood-pewee	1	1	0	1	0	0
Tennessee warbler	1	1	0	0	1	0
Solitary vireo	0	1	2	0	0	0
Golden crowned kinglet	0	0	0	0	2	1
Common grackle	2	0	0	0	0	0
Wilson's warbler	0	2	0	0	0	0
Yellow-bellied flycatcher	0	0	0	2	0	0

Table D4. Continued.

	Distance from streams (m)					
Bird Species	25	75	125	175	225	275
Ruby-crowned kinglet	0	0	0	1	0	1
American redstart	1	0	0	0	0	0
Black-billed cuckoo	1	0	0	0	0	0
Magnolia warbler	1	0	0	0	0	0
Northern oriole	1	0	0	0	0	0
Nashville warbler	0	0	1	0	0	0
Warbling vireo	0	0	1	0	0	0
White-breasted nuthatch	0	0	1	0	0	0
Yellow-rumped warbler	0	0	0	1	0	0
Northern flicker	0	0	0	1	0	0
Bay-breasted warbler	0	0	0	0	0	1
Total no. species	26	20	19	17	17	14
Cumulative no. species	26	31	36	40	42	43

APPENDIX E. NUMBER OF SIGHTINGS OF EACH SONGBIRD SPECIES IN UPLAND AND FLOODPLAIN FORESTS, MAINE, 1989-1990.

	Upland[a]		
Species	1989	1990	Floodplain[b] (1990)
Black-and-white warbler	78	48	23
Black-capped chickadee	52	34	24
Least flycatcher	17	9	19
Ovenbird	75	22	6
Veery	22	10	15
Golden-crowned kinglet	55	19	3
Red-eyed vireo	33	12	9
Black-throated green warbler	47	19	0
American robin	24	9	9
Northern parula	23	13	5
Common yellowthroat	13	1	16
Northern waterthrush	2	2	15
Red-breasted nuthatch	15	13	3
Black-throated blue warbler	24	15	0
Hermit thrush	28	14	0
Yellow-bellied sapsucker	19	6	6
Chestnut-sided warbler	5	3	7
Brown creeper	4	2	6
Canada warbler	17	3	5
Yellow warbler	2	0	8
Downy woodpecker	5	1	6
Great crested flycatcher	4	2	5
Blackburnian warbler	26	6	0
Eastern wood-pewee	5	3	3
Solitary vireo	11	3	3
Yellow-rumped warbler	12	5	1
Blue jay	16	5	0
Eastern kingbird	0	0	5
Magnolia warbler	21	4	1
Winter wren	15	5	0
American redstart	16	3	1

APPENDIX E. CONTINUED.

	Upland		
Species	1989	1990	Floodplain (1990)
Rose-breasted grosbeak	6	1	3
Swamp sparrow	0	0	4
Swainson's thrush	7	4	0
White-throated sparrow	14	0	4
Alder flycatcher	3	0	3
Cedar waxwing	6	0	3
Ruby-crowned kinglet	0	1	2
Red-winged blackbird	0	0	3
Tennessee warbler	0	0	3
Yellow-bellied flycatcher	1	1	2
Common grackle	7	0	2
Nashville warbler	5	1	1
Warbling vireo	0	1	1
Wilson's warbler	1	0	2
Black-billed cuckoo	0	0	1
Bay-breasted warbler	5	0	1
American crow	6	1	0
Northern flicker	3	0	1
Gray catbird	0	1	0
Hairy woodpecker	2	1	0
Northern oriole	1	0	1
Scarlet tanager	4	1	0
Song sparrow	1	1	0
White-breasted nuthatch	6	0	1
Wood thrush	0	1	0
Pine warbler	3	0	0
Pileated woodpecker	3	0	0
Purple finch	2	0	0
Eastern phoebe	1	0	0
Yellow-throated vireo	1	0	0
Evening grosbeak	1	0	0

[a] 1989: n = 21 upland sites; 1990: n = 10 upland sites

[b] n = 8 floodplain sites (1990)

Biography

Sarah T. Hooper was born in Flint, Michigan on 24 July 1964. She graduated from North Farmington High School in 1982.

Sarah earned a Bachelor of Arts degree in Biology from Earlham College in June 1986, graduating with departmental honors, college honors, and Phi Beta Kappa. For the following year, Sarah wandered the country, exploring the many things to do between the Atlantic and Pacific coasts. In 1987, Sarah worked in Arizona first for the Bureau of Land Management as a Student Conservation Association volunteer, and then for the Wyoming Cooperative Fish and Wildlife Research Unit as a biological technician, studying habitat use and movement patterns of four species of rails.

Sarah began graduate study at the University of Maine in April 1988, and was a graduate research assistant in the Department of Wildlife. She is a candidate for the degree of Master of Science in Wildlife Management in May 1991.