EFFECTS OF AGRICULTURE INTENSIFICATION ON LANDSCAPES AND AVIAN COMMUNITY STRUCTURE

By

JON CHARLTON BOREN

Bachelor of Science
Oklahoma State University
Stillwater, Oklahoma
1989

Master of Science
Oklahoma State University
Stillwater, Oklahoma
1992

Submitted to the Faculty of the
Graduate College of the
Oklahoma State University
in partial fulfillment of
the requirements for
the Degree of
DOCTOR OF PHILOSOPHY
December, 1995

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LANDSCAPES AND AVIAN COMMUNITY

STRUCTURE

Thesis Approved:

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Thesis Adviser

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Dean of the Graduate College

ACKNOWLEDGMENTS

I would like to express my sincere thanks and appreciation to my major advisor, Dr. David M. Engle, for his guidance and encouragement throughout this research project. His knowledge, experience, and dedication to research have been an invaluable part of my professional development. Additionally, I would like to thank my committee members Drs. Ronald E. Masters, Terrence G. Bidwell, David L. Nofziger, and Vernon A. Mast for their valuable assistance and interest in my education. I also would like to thank Mike Palmer and Mark Gregory for their technical guidance.

Special thanks go to my father and mother, Paul R., Sr. and Sue N. Boren for their continued moral support throughout my endeavors. Thanks for always being there for me.

Finally, I thank my wife Kristin who provided encouragement, unwavering love, and was a real believer in my abilities, and my son Jacob who has given me inspiration and endless joy. Thanks to both of you, I couldn't have done it without you.

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CHAPTER I

INTRODUCTION

This dissertation is composed of 3 manuscripts formatted for submission to selected scientific journals. Each manuscript is complete as written and does not require additional support material. The order of arrangement for each manuscript is text, literature cited, tables, and figures. Chapter II, "Landscape structure and change in a hardwood forest-tallgrass prairie ecotone of northern Oklahoma," is written in the format of the Journal of Range Management. Chapter III, "Effects of land use change on breeding bird community structure in a forest-grassland ecotone," is written in the format of the Journal of Wildlife Management. Chapter IV, "Prediction of vegetation cover type and avian species occurrence in a rural and urban-influenced landscape," is written in the format of the Journal of Applied Ecology.

CHAPTER II

LANDSCAPE STRUCTURE AND CHANGE IN A HARDWOOD FOREST-TALLGRASS PRAIRIE ECOTONE OF NORTHERN OKLAHOMA

Abstract

Temporal changes in land use, vegetation cover types, and landscape structure were examined in a hardwood forest-tallgrass prairie ecotone in northern Oklahoma using a Geographic Information System (GIS). Our objective was to examine relationships between urban sprawl, changes in land use and vegetation cover type, and landscape structure between 1966 and 1990. Most cover types and measures of landscape structure changed relatively little in the last 24 years. Most of the urban-influenced landscape in this study was managed for high-input agricultural products and resulted in a landscape with lower diversity, higher homogeneity, and greater patch fragmentation compared to the more rural landscape. Both native grasslands and forests were less fragmented in the rural landscape while forests were increasingly fragmented in the urban-influenced landscape. Native grasslands were less fragmented than forests for all years in both rural and urban-influenced landscapes. With increasing urban sprawl into rural landscapes, information on vegetation cover types and changes in landscape structure will better enable biologists to manage landscapes to preserve biological diversity.

Introduction

Increased agricultural development of the Great Plains since the 1870's has caused a dramatic decline in the aerial extent of tallgrass prairie. This decline exceeds any other major ecosystem in North America (Samson and Knopf 1994). Large, extensively managed blocks of rangeland and forests have been fragmented into smaller blocks of intensively managed introduced pastures and cropland through changing land use and altered ownership patterns. Urban sprawl into rural landscapes may further change land use and vegetation cover type and may operate to alter landscape structure and diversity. Land use changes often reduce ecosystem diversity on a regional scale by replacing natural vegetation with managed systems of altered structure (Krummel et al. 1987, McNeely et al. 1990). These anthropogenic changes have caused concern about preserving and managing for biological diversity (Grove and Hohmann 1992, Urban et al. 1992, West 1993). Urban development of wildlands tends to decrease biodiversity by increasing landscape homogeneity, dominance, and fragmentation (Davis and Glick 1978).

Although scientific literature contains extensive research on the effects of urbanization and fragmentation of contiguous forests, research is lacking in native grasslands (Samson and Knopf 1994) and grassland-forest ecotones (Risser 1990). Ecotones provide valuable insight to the complex dynamics of ecosystems including temporal changes in landscape structure and function (Wiens et al. 1985, Hardt and Forman 1989). Although ecotones are dynamic and typically have high community diversity (Risser 1990, Johnston et al. 1992), anthropogenic influences on change have not been well documented. Ecotones may also be used for early detection of global climatic change because many species are at the limits of their distributional ranges that are often climatically controlled (Hanson et al. 1992).

Changes in landscape structure may affect a wide variety of ecological processes (Turner 1989); but, relatively little is known about how components of landscape change over time (Baker 1992). Therefore, descriptions of changing landscape patterns form an important component of our understanding of ecological dynamics necessary to integrate the often conflicting demands of wildlife habitat, recreation, agriculture, and development. We used aerial photography from 1966 to 1990 in a hardwood forest-tallgrass prairie ecotone as the data set for addressing the relationships between urban sprawl and changes in land use, vegetation cover type, and landscape structure. We hypothesized that 1) land use, vegetation cover type, and landscape structure in a hardwood forest-tallgrass prairie ecotone have changed temporally in northern Oklahoma, 2) urban-influenced and rural areas differ in temporal change in land use, vegetation cover types, and landscape structure; and, 3) native grasslands have become more fragmented than forests because of increased human-intensive activities in native grasslands.

Study Site

Our study was centered around suburban Tulsa, Oklahoma, and the surrounding wildlands. The study included northeastern and southeastern Osage County, southern Washington County, and northern Tulsa County. The selection of the study area was based on areas with a suburban-wildland transition and areas with available historical aerial photoography. The study area included four U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) routes: 024 (Collinsville), 025 (Barnsdall), 026 (Bartlesville), and 125 (Skiatook) (Fig. 1). Legal description of these survey routes are provided by Baumgartner and Baumgartner (1992).

The BBS routes lie on an ecotonal area between the Cherokee Prairie grassland formation and oak-hickory savanna of the Cross Timbers (Bruner 1931, Soil Conservation Service 1981). The Cherokee Prairie of Oklahoma extends as a long, narrow strip, 240 km southward from the Kansas state line with a width ranging from 48 to 96 km throughout most of its length. The area is better adapted to support grasses than forests because of climate and underlying geology (Harlan 1957). The Cross Timbers of Oklahoma lie west of the Cherokee Prairie and the Lower Arkansas Valley, extending 288 km southward from Kansas with a width of 80 km. The region is a transitional oak forest with interspersed grasslands (Bruner 1931, Gray and Galloway 1959).

Survey routes varied in their proximity to Tulsa, a major metropolitan area in northern Oklahoma with a estimated population of 361,628 (U.S. Department of Commerce 1990). The Collinsville route is 24 km from Tulsa, in Washington County while the Bartlesville route is 74 km from Tulsa, in Osage County. Human population density of Washington and Osage County in 1990 was 3340 km^{-2} and 520 km^{-2}, respectively. We considered the Collinsville route to be subjected to more urban influence than the Bartlesville route. Thus, we viewed the Collinsville route as an intensively managed landscape and Bartlesville route as an extensively managed landscape, and from this point forward, the 2 landscapes will be discussed as urban-influenced or rural. Barnsdall and Skiatook routes are approximately 48 km from Tulsa and were selected as intermediate between urban-influenced and rural landscapes.

Methods

Data Collection

Black and white aerial photographs for 1966, 1973, 1980, and 1990 were obtained from the U.S. Department of Agriculture, ASCS, Aerial Photography Field Office, Salt Lake City, Utah. Photographs were 60.96 X 60.96-cm enlargements with a representative fraction (RF) of 1:7,920. We used portions of photography that covered BBS routes (40.2 km in length) and 0.8-km on each side of the route boundary. The resulting coverage was approximately 6430 ha for each route.

Topographic quadrangle maps, photo inspected in 1976, showed the natural and man-made features of the land at 1:24,000 scale and were obtained from the Oklahoma Geological Survey, Norman, Oklahoma. The quadrangles indicate both geographical coordinates and specific features such as vegetation, water, roads, and towns. These maps were used for both geo-registration of the photography and to aid in photo-interpretation.

Features identified on each photograph included: BBS route, roads, buildings and houses, land use, and vegetation cover types. Land use and vegetation cover types were interpreted based on the classification scheme of Stoms et al. (1983) (Table 1). All interpreted polygons of interest were traced on overlying acetate and supervised photo interpretation was compared to the 1990 photography.

Completed polygons were digitized using a digital scanner. Scanned images were edited, rectified, and vectorized using LTPlus (Line Trace Plus, version 2.22) and imported into the GIS GRASS (Geographic Resource Analysis Support System) (Shapiro et al. 1992). Vector maps were then patched together to form the complete route, labeled, and converted to a raster map with 5-m resolution.

Data Analysis

Temporal changes in land use and vegetation cover types were examined using GIS to describe differences between and within routes for the 24-yr period. Landscape analysis was performed using the raster landscape ecological (r.le) spatial analysis package within GRASS (Baker and Cai 1992), which was developed for quantitative analysis of landscape structure. The r.le programs were used to generate landscape measures of mean patch size, fractal dimension, richness, Shannon diversity, dominance, contagion, angular second moment, and contrast.

Mean patch size is the mean area (ha) of patches in the sampling area and serves as an index of fragmentation. It is calculated for all patches in the sampling area by dividing sample area size by the number of patches (Baker and Cai 1992). As patches become smaller because of fragmentation, mean patch size decreases. Fractal dimension is a measure of fractal geometry or patch shape complexity of a landscape (Mandelbrot 1983, Krummel et al. 1987). Fractal dimension (F) was calculated by regressing polygon area against perimeter length for each landscape patch. Values for fractal dimension range from 1 to 2. Landscapes dominated by simple patterns (circles and squares) have low values of F while landscapes dominated by complex or convoluted patterns have high values of F (Krummel et al. 1987).

Shannon's diversity index (H) combines richness and evenness. Richness refers to the number of patch attributes present in the sampling area and evenness refers to the distribution of area among different patch types (Turner 1990a, 1990b). Richness and evenness are the compositional and structural components of diversity, respectively (McGarigal and Marks 1994). The larger the value of H, the more diverse the landscape (O'Neill et al. 1988). The dominance index (D) is based on the Shannon-Weaver diversity index (Shannon and Weaver 1962) but emphasizes deviation from evenness.

The dominance index measures the extent that specific land uses (or vegetative cover types) dominate the landscape (O'Neill et al. 1988). Large values of D indicate a landscape dominated by one or a few cover types while low values of D indicate a landscape with many cover types represented in approximately equal proportions (Turner 1990a).

Three texture measures were calculated for the regional landscape which included contagion, angular second moment, and contrast using eight-neighbor analysis to quantify the adjacency of similar patch types. Contagion (C) measures the extent to which cover types are aggregated or clumped in contiguous patches (O'Neill et al. 1988). A landscape with well interspersed patch types will have a lower contagion compared to a landscape with poorly interspersed patch types (McGarigal and Marks 1994). Angular second moment is a measure of landscape homogeneity. Larger values for angular second moment indicate more homogeneity (McGarigal and Marks 1994). Contrast measures local variation present in the landscape (Baker 1994).

Temporal changes in land use, vegetation cover types, and landscape structure for the region were analyzed by comparing the mean of all 4 routes over all years. Comparisons also were made with 1900 data from Criner (1995) for the mean of the Collinsville and Bartlesville routes to investigate historical changes in cover types and landscape structure for the region. In addition, comparisons between 1966 and 1990 were made between Collinsville and Bartlesville to assess the effects of urbanization on vegetation cover types and landscape structure. Mean patch size and fractal dimension were also determined for native grasslands and forests within both Collinsville and Bartlesville to assess the effects of urban influences on fragmentation in these landscape types.

Results and Discussion

Temporal Change in the Region

Cover Types

Developed areas of the northern Oklahoma landscape increased by 27% from 1966 to 1990 (Fig. 2). In comparison, there was a 3,270% increase in developed areas between 1900 and 1990. The increase in developed areas altered both vegetation cover types and landscape structure. Cropland decreased from 1966 to 1990 by 84% (Fig. 2). In comparison, total area of cropland decreased by 34% between 1900 and 1966. Reduction in cropland in recent years may be the result of continued decrease in cultivation of marginal lands (Sampson and Knopf 1994).

Decline of native grassland in our study area was well below national estimates. Declines in tallgrass prairie have been estimated at 82% to 99% in North America (Sampson and Knopf 1994). However, native grasslands declined only 36% between 1900 and 1990. The rate of loss has greatly decreased in recent years. Native grassland declined from 1,217 ha to 1,178 ha between 1966 and 1990 (Fig. 2). This suggests losses of tallgrass prairie are not widespread in this area and losses are lower on areas with marginal cropland, as in the hardwood forest-tallgrass prairie of Oklahoma, than in the tallgrass prairie region in general.

From 1900 to 1966, a 22% expansion in area of forest into native grassland was observed along with a concomitant 36% decrease in native grasslands, which may have been related to the absence of brush treatment practices (primarily herbicides) (Rollins 1987). An 87% increase in brush-treatment lands since 1966 resulted in reduction of forests from 34% of the area in 1966 to 27% of the area in 1990. In comparison, forest cover in 1900 was 29% of the area resulting in a difference of only 8% between 1900 and 1990. Bare ground decreased by 44% between 1966 and 1990, probably because of the 82% reduction in oil and gas activity.

Landscape Structure

Landscape structure can be characterized by the composition and relative abundance of vegetation cover types and their spatial arrangement or geometry (Freemark et al. 1993). Natural and anthropogenic disturbances alter landscape structure and may have important ecological implications (Turner 1990b). Therefore, temporal changes in landscape structure must be considered in quantitative landscape studies (Dunn et al. 1990). Temporal changes observed in land use and vegetation cover types in our study resulted in altered landscape structure.

Fragmentation occurs when areas of homogeneous habitat are broken into a mosaic of smaller, dissimilar habitat patches (Laudenslayer 1984, Temple and Wilcox 1984). Mean patch size declined only 16% between 1966 and 1990 (Table 2). In comparison, between 1900 and 1966 mean patch size declined 77%. More local variation between patch types was also observed (angular second moment and contrast) between 1900 and 1966. Although landscape fragmentation and degree of heterogeneity have generally stabilized in recent years, the increase in local variation between patch types continued from 1966 to 1990 (Table 2). Patch complexity (i.e., fractal dimension), between 1966 and 1990, changed at a slower rate (0.1 per year) than between 1900 and 1966 (0.2 per year).

A shift from native to more intensively managed landscapes generally results in a larger proportion of the landscape being dominated by fewer cover types (Laudenslayer 1984). However, little change in landscape diversity and contagion occurred between 1966 and 1990 (Table 2). Landscape dominance decreased by 15% between 1900 and

1966 indicating a landscape dominated by fewer cover types, primarily native grasslands, which comprised the majority of the landscape in 1900. The change in landscape dominance supports the observed 54% increase in Shannon diversity from 1900 to 1966 (0.87 and 1.34, respectively). An 11% increase in contagion also was observed from 1900 to 1966 indicating a landscape with more highly interspersed patches of land cover types and uses in 1900. In contrast, landscape dominance increased by 10% between 1966 and 1990 (Table 2) and approached its historical value indicating a landscape dominated by fewer land uses or cover types in 1990.

Effects of Urbanization

Temporal Changes in Cover Types

A 50% increase in developed areas was observed in the urban-influenced route (Collinsville) while a 4% decrease was observed in the rural route (Bartlesville) between 1966 and 1990 (Fig. 3 and 4). Therefore, these routes provided excellent study areas for investigation of the effects of urbanization on vegetation change and landscape structure. Because the urban-influenced route was located near Tulsa, it experienced a greater amount of human influx (i.e., urban sprawl) over the past 25 years compared to the rural route and resulted in different temporal changes in vegetation cover types.

The urban-influenced route was subject to more intensive management practices, such as cropland and pasture land and hay meadows than the rural route. Cropland accounted for 17% of the urban-influenced route and only 1% of the rural route in 1966. Both routes had a reduction in cropland between 1966 and 1990; however, the rate of loss in cropland was greater for urban-influenced route (78%) compared to the rural route (48%) (Figs. 3 and 4). Cropland in the urban-influenced route was converted primarily to pasture land and hay meadows. Pasture land and hay meadows accounted for 21% of the urban-influenced route and only 3% of the rural route in 1966. In addition, pasture land and hay meadows increased by 48% in the urban-influenced route but decreased by 46% in the rural route (Figs. 3 and 4). The increase in pasture land and hay meadows in the urban-influenced route resulted from the conversion of native grassland, cropland, and forests. This suggest the urban-influenced route was managed for high-input agricultural products compared to a rural route.

Deciduous forests accounted for 37% of the rural route and only 13% of the urban-influenced route in 1966. Forests were converted primarily to brush-treated lands in the rural route and to pasture land and hay meadows in the urban-influenced route from 1966 to 1990. However, the rate of decline in forest was greater for the rural route than the urban-influenced route because of differences in the amount of brush-treated lands between the routes. Brush-treated lands accounted for 12% of the rural route and only 1% of the urban-influenced route in 1966. In addition, there was a 121% increase in brush-treated lands from 1966 to 1990 in the rural route while the urban-influenced route showed relatively little change (Figs. 3 and 4). Native grasslands were the dominant cover type for both routes in all years (Figs. 3 and 4). However, there was very little change in native grasslands in the urban-influenced route while a 19% decline was observed in the rural route from 1966 to 1990. The decline in native grasslands along the rural route may be misleading because native grasslands subjected to herbicides or fire were photo-interpreted as brush-treated lands. Maintenance of tallgrass prairie dominance in this region requires fire or herbicides to prevent encroachment of woody species (Bragg and Hulbert 1976, Knight et al. 1994).

Temporal Changes in Landscape Structure

Mean patch size is generally large in areas of natural vegetation influenced minimally by human activities (Pickett and Thompson 1978). With increased human activity, mean patch size decreases because the landscape is generally subdivided into smaller patches (Forman and Boerner 1981). Measures of mean patch size in our study indicated the urban-influenced route became more fragmented than the rural route since 1973 (Table 3). In addition, mean patch size declined by 29% in the urban-influenced route and only 7% in the rural route from 1966 to 1990 suggesting landscape fragmentation was 4 times greater in the urban-influenced route. Human activities related to crop production and urban development also tend to simplify patch shapes resulting in lower fractal dimensions (Krummel et al. 1987, O'Neill 1988). However, patch complexity between routes was similar and slightly increased in both routes since 1973 (Table 3). This suggests natural disturbance regimes including climate in the landscape may have influenced patch complexity to a larger degree than human activities.

Urbanization typically decreases diversity by increasing landscape fragmentation, homogeneity, and dominance (Davis and Glick 1978). Landscape dominance increased by 32% in the urban-influenced route (Table 3) suggesting a general trend for the landscape to be dominated by fewer land uses or vegetation cover types (O'Neill et al. 1988). Landscape dominance decreased by 11% in the rural route suggesting a general trend toward land uses or vegetation cover types represented in more equal proportions. In addition, angular second moment increased by 19% from 1966 to 1990 in the urban-influenced route suggesting a homogeneous, less diverse landscape. In contrast, angular second moment decreased by 17% from 1966 to 1990 in the rural route indicating a landscape becoming more heterogeneous suggesting an increase in landscape diversity. Although landscape diversity was 15% greater in the urban-influenced route compared to the rural route in 1966, landscape diversity declined by 11% in the urban-influenced route while landscape diversity increased by 8% in the rural route from 1966 to 1990 (Table 3). Overall, the urban-influenced landscape is becoming less diverse, but the rural landscape is becoming more diverse since 1966.

Fragmentation of Native Grasslands and Forests

The Collinsville and Bartlesville routes were used to investigate fragmentation of native grasslands and forests in a urban-influenced and rural environment since 1966. Urbanization tends to simplify patch complexity and increase fragmentation of contiguous forests (Godron and Forman 1983). Forest patches were more complex in shape compared to native grasslands in the urban-influenced route (Table 4), suggesting there was more human impact and fragmentation in native grasslands than forests. However, native grassland fragmentation remained relatively unchanged while forest fragmentation increased by 26% from 1966 to 1990 (Table 4). Therefore, relationships between urbanization, patch complexity, and fragmentation in other ecosystems such as contiguous forests may not always be appropriate in the forest-tallgrass prairie ecotone.

Native grassland patches were more complex in shape compared to forest patches in the rural route for all years (Table 4) which we attribute to fire and other brush treatment practices. Disturbance patches created by prescribed burning can increase landscape heterogeneity and patch complexity because fire effects differ with respect to topography, fuel type, fuel load, climate, and season (Godron and Forman 1983, Biondini et al. 1989, Baker 1992, Urban 1994). In addition, fragmentation of both native grasslands and forests in the rural route decreased 57% and 37% respectively from 1966 to 1990.

In both routes there was relatively little change in complexity of patch shape (fractal dimension) in either native grasslands or forests over time (Table 4). In addition, native grasslands were less fragmented than forests for all years based on mean patch size in both routes. Because native grasslands were less fragmented than forests one would expect to find increased road and residential growth in the forests compared to the native grasslands. However, our data indicated that roads were developed randomly with respect to cover type in the landscape. Human impact areas, including residential development, were primarily located in native grasslands in the urban-influenced route in 1966 (Table 5). However, with the temporal increase in urbanization, forests were increasingly selected for human development from 1973 to 1990 (Table 5). This may account for the observed temporal increase in fragmentation of forests. However, in the rural route human impact areas were more evenly distributed between native grasslands and forests for all years (Table 5). Historically, forests were more fragmented than native grasslands for both routes (Criner 1995). Therefore, observed differences in fragmentation between cover types is most likely a function of geomorphologic processes such as soils and natural disturbance regimes including climate and fire (Godron and Forman 1983).

Conclusions

We found areas surrounding urban centers eroded in landscape quality, as defined by landscape fragmentation and diversity. In contrast, landscape quality improved in the rural areas dominated by ranching enterprises. Differences in landscape quality between landscapes (urban-influenced vs. rural) can be attributed to differences in land use and associated management practices. Maintenance of the tallgrass prairie by prescribed burning, herbicide application, and grazing management most likely accounts for the observed improvement in landscape quality in rural (extensively managed) areas while the temporal increase in seeded pasture land and hay meadows accounts for the observed reduction in landscape quality in urban-influenced (intensively managed) areas. Therefore, these and other similar landscapes will continue to diverge in landscape quality in the absence of societal pressure to halt urban sprawl into rural landscapes.

Our analysis suggests biologists and conservationist should focus their concerns on fragmentation and biological diversity in urban and suburban ecosystems. Wildlands will continue to be fragmented and altered by urban sprawl, which creates the most noticeable change in a landscape (Gates 1991). Therefore, human impact on ecological systems as a result of changing land use is a critical problem facing biologists today. For example, changes in wildlife diversity and density by urbanization result from human-induced changes in vegetation composition and structure. With increasing urbanization and suburban sprawl into rural landscapes, information on vegetation cover types and changes in landscape structure will better enable biologists to manage landscapes to conserve biological diversity (Grove and Hohmann 1992). Line, strip, and stream corridors also must be an integral part of the urban planning process to help offset the effects of fragmentation. By the year 2000, 90% of the United States population is projected to live in urban and suburban environments (George 1982) which further necessitates the need to understand the complex relationships between urban sprawl and changes in land use, vegetation cover types, and landscape structure on ecological systems in urban environments.

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Table 1. Classification system used to map land use and vegetation cover types (adapted from Stoms et al. 1983).

Land use and cover type	Description
Developed area	Land occupied by residential, industrial, or other human structures and non-agricultural activities. Also includes transportation and utility facilities.
Roads	Black top, gravel, dirt roads and driveways
Water	Ponds, lakes, streams, and rivers
Cropland	Land cultivated for row crops and cereal grains but excluding grazing lands
Pasture land and hay meadows	Includes pasture land (seeded, grasslands used for grazing by cattle, sheep, goats, and horses) and hay meadows
Native grassland	Native grasslands with less than 10% cover by shrubs or trees
Deciduous forest	Vegetation dominated (>10%) by cover of broadleaf hardwoods. Mostly post oak (Quercus stellata) and blackjack oak (Q. marilandica)
Brush-treated land	Native vegetation subjected to herbicides, fire, or chaining to control woody brush encroachment
Bare ground	Land with less than 5% vegetative cover

Table 2. Measures of landscape pattern ($\underline{x}$ ± SE) and percent change from 1966 of the mean of four 25 mile transects in a hardwood forest-tallgrass prairie ecosystem in northern Oklahoma for 1900[1], 1966, 1973, 1980, and 1990.

Index	Year					% Change
	1900	1966	1973	1980	1990	1966
Mean patch size (ha)	18.01 ± 1.34	4.18 ± 0.33	3.97 ± 0.33	3.73 ± 0.31	3.50 ± 0.33	- 16.3
Fractal dimension	1.11 ± 0.07	1.25 ± 0.01	1.24 ± 0.01	1.26 ± 0.01	1.28 ± 0.01	+ 2.4
Shannon diversity	0.87 ± 0.08	1.34 ± 0.05	1.39 ± 0.03	1.28 ± 0.03	1.28 ± 0.05	- 4.5
Dominance	0.92 ± 0.08	0.78 ± 0.06	0.76 ± 0.01	0.86 ± 0.04	0.86 ± 0.05	+ 10.3
Contagion	2.55 ± 0.08	2.82 ± 0.07	2.82 ± 0.02	2.89 ± 0.04	2.88 ± 0.07	+ 2.1
Angular second moment	0.47 ± 0.01	0.29 ± 0.02	0.28 ± 0.01	0.31 ± 0.01	0.30 ± 0.02	+ 3.4
Contrast	0.16 ± 0.03	0.38 ± 0.03	0.43 ± 0.04	0.43 ± 0.05	0.46 ± 0.03	+ 21.1

[1]Data is from Criner (1995).

Table 3. Measures of landscape pattern and percent change from 1966 of urban-influenced and rural routes in a hardwood forest-tallgrass prairie ecosystem in northern Oklahoma for 1966, 1973, 1980, and 1990.

Index	Year				% Change
	1966	1973	1980	1990	
Urban-influenced (Collinsville)					
Mean patch size (ha)	4.16	3.93	3.22	2.96	- 28.8
Fractal dimension	1.23	1.25	1.27	1.28	+ 4.1
Shannon diversity	1.43	1.39	1.33	1.28	- 10.5
Dominance	0.65	0.75	0.81	0.86	+ 32.3
Contagion	2.69	2.83	2.85	2.91	+ 8.2
Angular second moment	0.27	0.30	0.30	0.32	+ 18.5
Contrast	0.33	0.46	0.50	0.50	+ 51.5
Rural (Bartlesville)					
Mean patch size (ha)	3.96	4.29	3.63	3.42	- 7.5
Fractal dimension	1.27	1.24	1.27	1.30	+ 2.4
Shannon diversity	1.21	1.29	1.29	1.31	+ 8.3
Dominance	0.93	0.78	0.78	0.83	- 10.8
Contagion	2.99	2.82	2.81	2.88	- 3.7
Angular second moment	0.35	0.30	0.29	0.29	- 17.1
Contrast	0.41	0.35	0.35	0.42	+ 2.4

Table 4. Measures of landscape pattern and percent change from 1966 of native grassland and forest of urban-influenced and rural routes in a hardwood forest-tallgrass prairie ecosystem in northern Oklahoma for 1966, 1973, 1980, and 1990.

Index	Year				% Change
	1966	1973	1980	1990	
Urban-influenced (Collinsville)					
Native grassland					
Mean patch size (ha)	15.24	15.37	12.52	15.25	+ 0.1
Fractal dimension	1.25	1.27	1.26	1.25	0.0
Forest					
Mean patch size (ha)	2.72	1.75	1.69	2.01	- 26.0
Fractal dimension	1.36	1.39	1.36	1.35	- 0.7
Rural (Bartlesville)					
Native grassland					
Mean patch size (ha)	11.83	9.81	12.81	18.55	+ 56.9
Fractal dimension	1.35	1.29	1.35	1.41	+ 4.4
Forest					
Mean patch size (ha)	3.99	6.21	4.43	5.47	+ 37.0
Fractal dimension	1.25	1.24	1.24	1.31	+ 4.8

Table 5. Percentage of native grassland and forest on a relative basis adjacent to human impact areas for the urban-influenced and rural routes for 4 separate years.

Cover type	Year			
	1966	1973	1980	1990
Urban-influenced (Collinsville)				
Native grassland	25	28	50	48
Deciduous forest	13	7	35	38
Rural (Bartlesville)				
Native grassland	36	34	44	47
Deciduous forest	40	46	32	47

Fig. 1. The 4 U.S. Fish and Wildlife Service Breeding Bird Survey routes located in northern Oklahoma used for the study area.

Breeding Bird Survey Routes

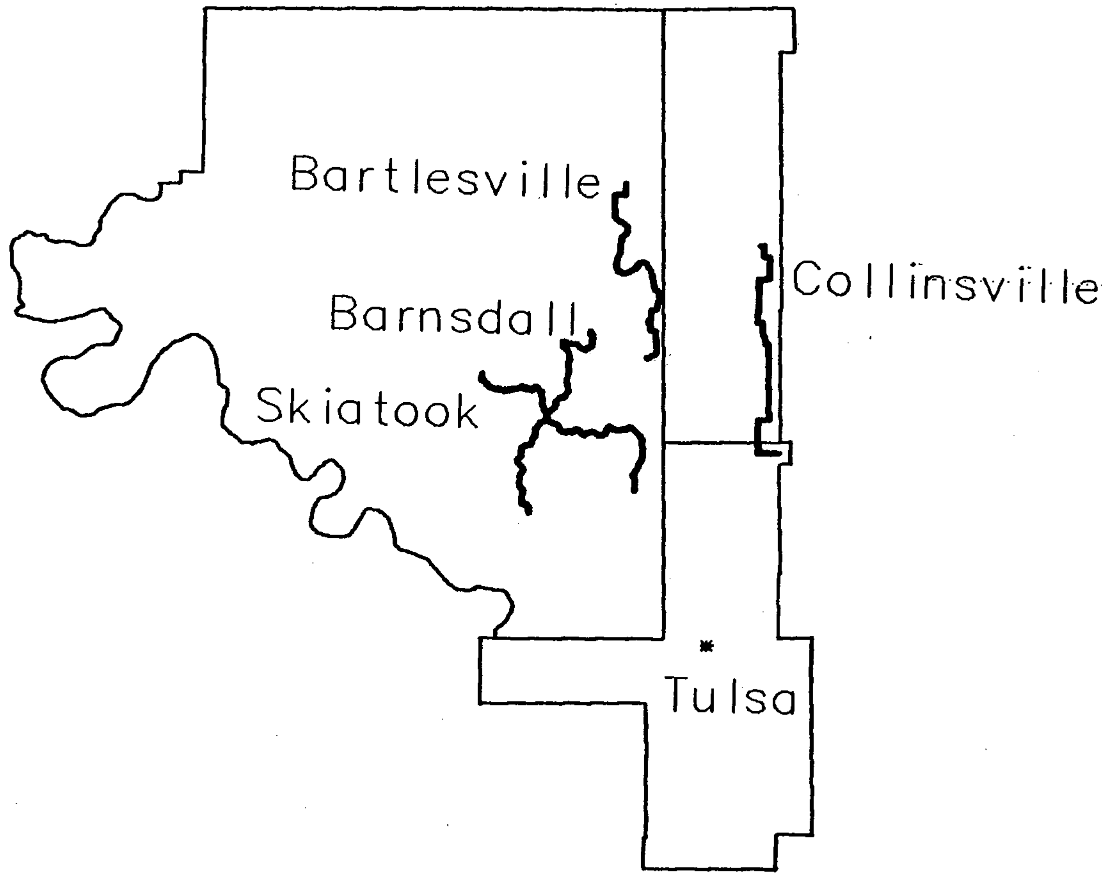
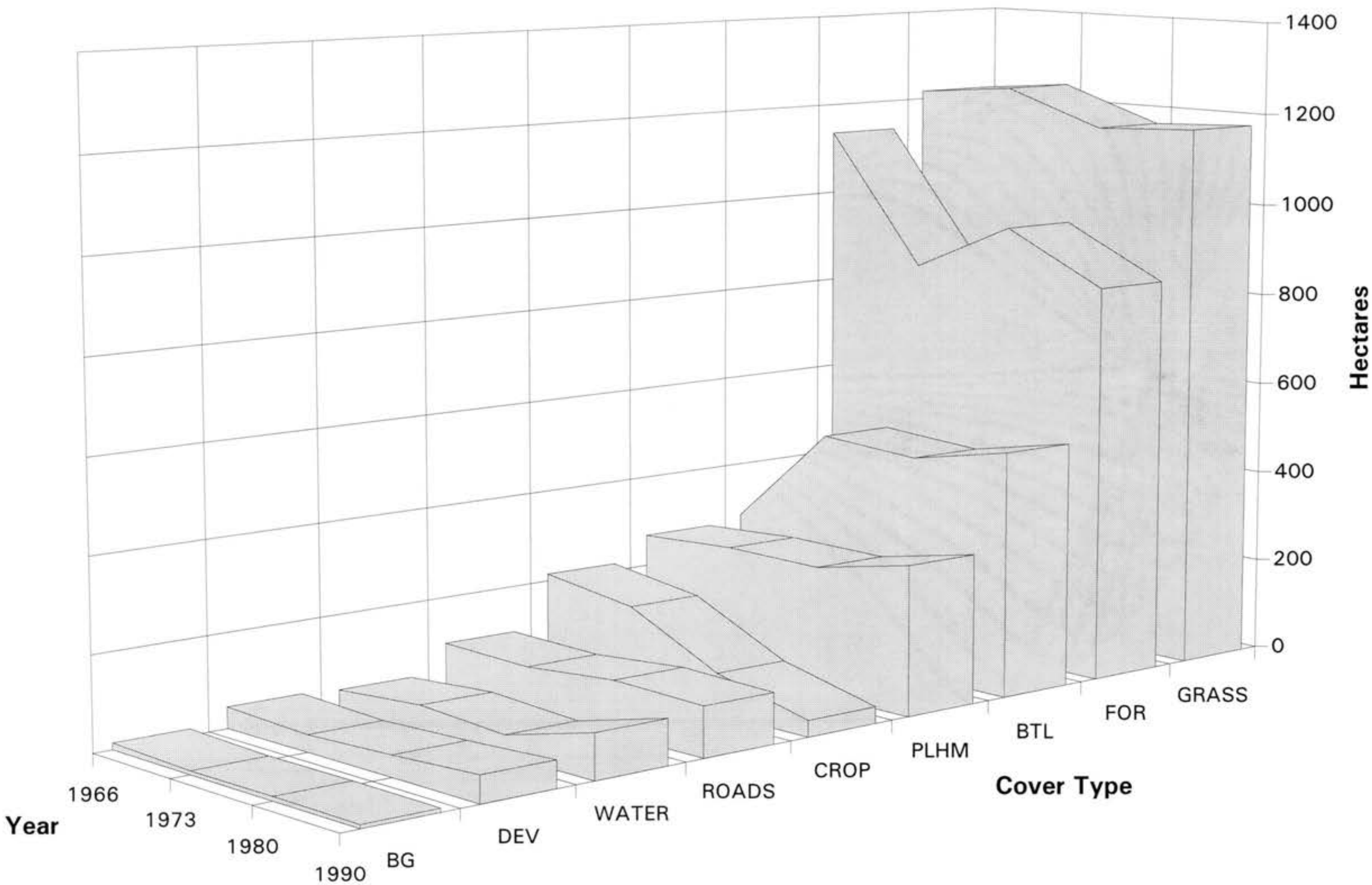


Fig. 2. Temporal changes in bare ground (BG), developed area (DEV), water, roads, cropland (CROP), pasture land and hay meadows (PLHM), brush-treated land (BTL), deciduous forest (FOREST), and native grassland (GRASS) of the mean of four 25 mile transects in a hardwood forest-tallgrass prairie ecosystem in Northern Oklahoma for 1966, 1973, 1980, and 1990.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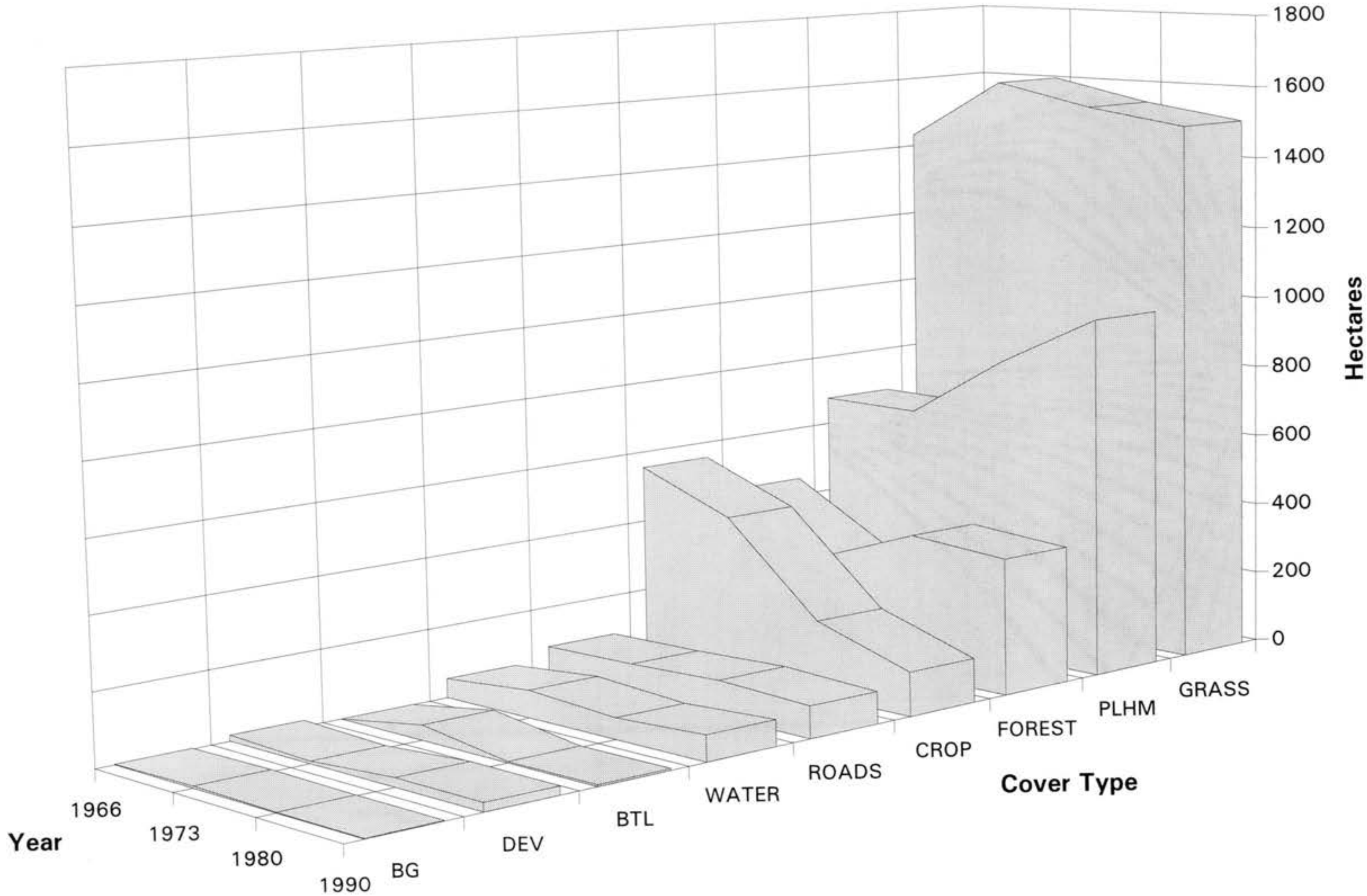
BTL
FOR
GRASS
Cover Type

Fig. 3. Temporal changes in bare ground (BG), developed area (DEV), brush-treated land (BTL), water, roads, cropland (CROP), deciduous forest (FOREST), pasture land and hay meadows (PLHM), and native grassland (GRASS) in the urban-influenced route (Collinsville) for 1966, 1973, 1980, and 1990.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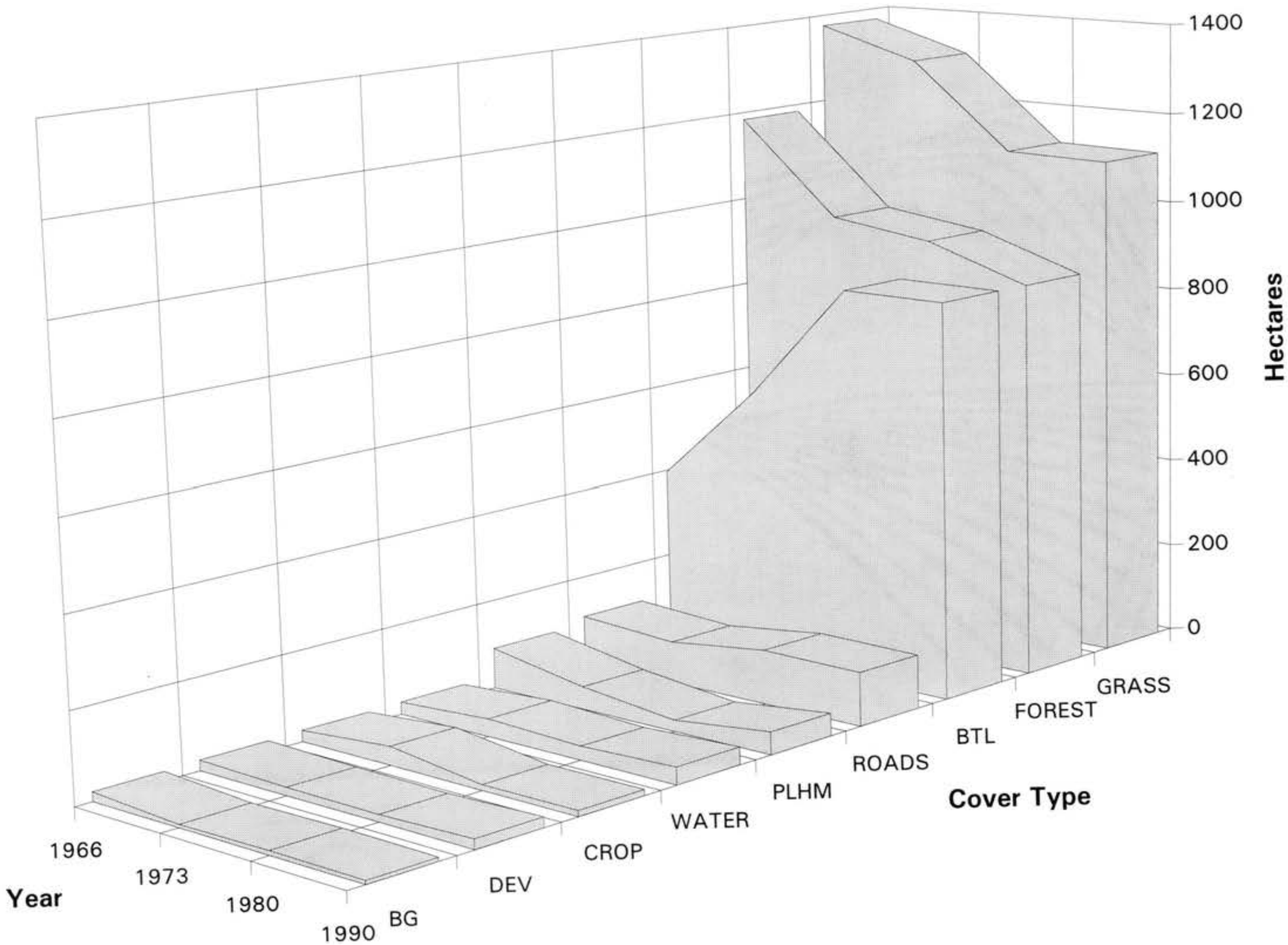
PLHM
GRASS

Fig. 4. Temporal changes in bare ground (BG), developed area (DEV), cropland (CROP), water, pasture land and hay meadows (PLHM), roads, brush-treated land (BTL), deciduous forest (FOREST), and native grassland (GRASS) in the rural route (Bartlesville) for 1966, 1973, 1980, and 1990.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CHAPTER III

EFFECTS OF LAND USE CHANGE ON BREEDING BIRD COMMUNITY STRUCTURE IN A FOREST-GRASSLAND ECOTONE

Abstract: We identified land uses, vegetation types, and landscape patterns associated with avian community diversity in a rural and urban-influenced landscape. We obtained long-term (24 years) changes in biological diversity of game bird and songbird community structure through records from the North American Breeding Bird Survey. We obtained historical and present land use, vegetation cover types, and landscape pattern of both landscapes from high-resolution aerial photography. We found that certain aspects of avian community structure are a function of the complex interaction of land use, vegetation cover type, and landscape pattern. Avian community structure is explained by different sets of environmental variables for the 2 landscapes. Changes in vegetation cover type altered avian community structure by decreasing some forest associated species in both landscapes relative to prairie and generalist species in the rural and urban-influenced landscapes, respectively.

Land use changes often reduce ecosystem diversity on a regional scale due to the replacement of natural vegetation with managed systems of altered structure (Davis and Glick 1978, Krummel et al. 1987, McNeely et al. 1990). These anthropogenic changes have caused concern about preserving and managing biological diversity (Grove and Hohmann 1992, Urban et al. 1992, West 1993). Management of avian diversity in urban environments has become increasingly important because of increasing urbanization, growth in non-consumptive uses, and economic returns of urban wildlife (Gill and Bonnett 1973, DeGraaf and Payne 1975, Smith 1975, George 1982). Although the effects of urbanization on many wildlife species are well known, the dynamics of heterogeneous environments, such as the wildland to suburban transition, have been largely ignored by ecology. Urban sprawl into rural landscapes of the Great Plains results in altered land ownership patterns and management practices that compound changes in land use, resulting in contrasting vegetation cover types and landscape structure between rural and urban-influenced landscapes (Boren 1995). As the human population continues to expand, more emphasis will be placed on maintaining avian biodiversity in order to protect desirable species (Rodiek 1991). However, few studies have compared the avifauna and vegetation of urban areas with the outlying natural areas (Beissinger and Osborne 1982).

Noss (1983) suggested that birds are useful as ecological indicators of biodiversity. Avian species are excellent candidates for scientific investigation at the landscape scale since they are habitat-specific. Robinson and Holmes (1984) and Rotenberry (1985) demonstrated that a strong association between individual bird species and vegetation cover type exists in grassland communities at the local population level. However, habitat selection by birds may be more a function of vegetation structure than floristic composition at the landscape level (Engstrom and

James 1981, Cody 1985, DeGraaf 1991). The relationship between vegetation cover types, habitat structure, and bird species composition is useful for examining the effects of land use on breeding birds at both stand and landscape scales and must be addressed when assessing habitat quality (DeGraaf 1991, Scott et al. 1993). However, most population-surveys, including both species richness and evenness, on non-game and many game species have been at spatial scales of approximately 40 ha in areas representing a single plant community (Urban and Shugart 1984). Therefore, habitat management to maintain high historical diversity of avian species is dependent on the knowledge of changes that can or will occur in a given landscape because the landscape is a mosaic of stands and local ecosystems (DeGraaf 1991).

Implications of urbanization on avian diversity and density in the hardwood forest-tallgrass prairie ecotone must largely be extrapolated from previous studies conducted in contiguous forests (Johnson and Temple 1986). However, native birds in North America's prairies have undergone more widespread declines over the past 25 years than any other U.S. bird group which warrants the increasing concern for the conservation of these birds (Knopf 1994). Therefore, we investigated the complex relationships between urban sprawl and changes in land use, vegetation cover types, and landscape structure on avian community structure. Our premise was that avian community structure differed between a rural and urban-influenced landscape in a hardwood forest-tallgrass prairie ecotone. Based on this premise, we hypothesized that 1) avian community in the urban-influenced and rural landscapes differed in 1966 and diverged over time as the urban-influenced landscape became more urbanized; and, 2) different landscape cover types, in part reflecting human activities, between the landscapes influenced avian community structure.

STUDY AREA

Our study was centered around suburban Tulsa, Oklahoma, and included surrounding wildlands in northeastern Osage and southern Washington counties. The selection of the study area was based on areas with a suburban-wildland transition and areas in which biological diversity of bird species were available. The study area included 2 U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) routes: 024 (Collinsville) and 026 (Bartlesville). Legal description of these survey routes are provided by Baumgartner and Baumgartner (1992).

The BBS routes lie on an ecotonal area between the Cherokee Prairie grassland formation and oak-hickory savanna of the Cross Timbers (Bruner 1931, Soil Conservation Service 1981). The Cherokee Prairie of Oklahoma extends as a long, narrow strip, 240 km southward from the Kansas state line with a width ranging from 48 to 96 km throughout most of its length. The area supports grasses, forbs, and legumes better than forests because of climate and underlying geology (Harlan 1957). The Cross Timbers of Oklahoma lie west of the Cherokee Prairie and the Lower Arkansas Valley, extending 288 km southward from Kansas and approximately 80 km wide. The region is a transitional oak forest with interspersed prairie (Bruner 1931, Gray and Galloway 1959).

Survey routes also varied in their proximity from Tulsa, a major metropolitan area in northern Oklahoma with a estimated population of 361,628 (U.S. Department of Commerce 1990). The Collinsville route is 24 km from Tulsa, in Washington County and the Bartlesville route is 74 km from Tulsa, in Osage County. A 50% increase in human use areas was observed in the Collinsville route while a 4% decrease was observed in the Bartlesville route between 1966 and 1990 (Boren 1995). Human population density of Washington and Osage County in 1990 was 3340 km^{-2} and 520 km^{-2}, respectively.

We considered the Collinsville route to be subjected to more urban influence than the Bartlesville route. Thus, we viewed the Collinsville route as an intensively managed landscape and Bartlesville route as an extensively managed landscape, and from this point forward, the 2 landscapes will be discussed as urban-influenced or rural.

METHODS

Bird Surveys and Database Construction

We utilized BBS routes from the U.S. Fish and Wildlife Service to obtain our avian diversity data. The BBS data set is the only data set that indexes the population status of many species of birds over a large geographical area and time (Bystrak 1981, Geissler and Noon 1981). Although a roadside count misses some species and is limited by road placement, the results are considered to be fairly reliable indexes for a prairie-woodland ecosystem (Baumgartner and Baumgartner 1992).

We classified avian species as neotropical migrants, temperate migrants, and residents, and grouped species into 5 designations of habitat occurrence: forest, forest edge and shrubland, prairie, wetland, and developed areas. We further grouped species into foraging zones: aerial (open zones), ground and shrub (foliage 0 - 3 m), midstory (foliage 3 - 10 m), canopy (foliage > 10 m), bole (trunks and limbs), and water. Nesting zones included ground, shrub (0 - 3 m), midstory (3 - 10 m), canopy (> 10 m), cavity, and other (variable heights and substrates).

Bird abundances, available from 1967 to 1991, were segregated around 4 years (1966, 1973, 1980, and 1990) for which landscape cover type and structure data were documented previously for both landscapes (Boren 1995). Thus, breeding bird data from 1967 to 1970 corresponded to the 1966 landscape data, BBS data from 1971 to 1976 corresponded to the 1973 landscape data, BBS data from 1977 to 1984 corresponded to the 1980 landscape data, and BBS data from 1985 to 1991 corresponded to the 1990 landscape data. Relative abundance was then calculated for each of the 4 time periods by averaging relative abundance for the 4 years. Landscape data included land use and vegetation cover types (Table 1) and landscape structure measures included mean patch size, fractal dimension, landscape richness, Shannon diversity, dominance, contagion, and angular second moment (Boren 1995).

Data Analysis

Avian Community Change.--We performed detrended correspondence analysis (DCA) with the program CANOCO (ter Braak 1988) to determine if avian community structure differed between landscapes and to document shifts in avian community structure over time by using year as the passive environmental variable. Detrended correspondence analysis is an indirect gradient analysis in which samples (species abundances) are arranged according to species composition alone. The important environmental gradients are indirectly inferred from the trends in species composition. The first 2 axes of the DCA ordination were selected as the main ordination framework because higher eigenvalues indicate more importance in explaining avian community variability (Table 2). Detrended correspondence analysis has the advantage of producing axes that correspond to actual ecological distances, as defined by the abundance of species, and are not forced to be equal in length (Malanson and Trabaud 1987). We plotted the centroids for avian community structure for individual years in DCA space as points. We used these points to indicate trajectories through time in the avian space defined by the ordination axes (Whisenant and Wagstaff 1991).

We used species scores generated by DCA to determine the avian species responsible for temporal shifts in avian community structure. Visual observation of axis 1 and 2 of the ordination diagram indicated bird species (with overall abundances greater than 3) most responsible for temporal change in avian community composition.

Therefore, DCA provided a multivariate approach to the identification of declining or increasing species within each landscape.

Influence of Landscape Cover Type and Structure.--We performed canonical correspondence analysis (CCA) with the program CANOCO (ter Braak 1988) to determine the influence of landscape cover type and structure on the breeding bird community for each landscape. Canonical correspondence analysis is an eigenvector ordination technique for multivariate direct gradient analysis (ter Braak 1986). This technique explains community variation by detecting patterns of variation in species abundance that can best be explained by a set of environmental variables (ter Braak 1986). By applying CCA it is possible to identify important environmental variables that explained community structure with no a priori knowledge about possible predictor variables (Saetersdal and Birks 1993).

We related abundances of all bird species in urban-influenced and rural landscapes (100 and 86 bird species, respectively) to both landscape cover type and structure variables in separate CCA ordinations. We used forward selection and Monte Carlo permutation tests ($P < 0.05$) to determine environmental variables that best explained variation in breeding bird abundances. We examined canonical coefficients and intraset correlations to evaluate relative contributions of environmental variables to the axes. We also used unrestricted Monte Carlo permutation tests to test statistical significance ($P < 0.05$) of the first 2 ordination axes. Tests of significance in CCA do not depend on parametric distributional assumptions; therefore, we did not transform species and environmental variables (Palmer 1993).

Canonical correspondence analysis biplots provided weighted least squares approximations of the weighted averages of species identified as causing shifts in community structure (from DCA) with respect to environmental variables (ter Braak

1986). We examined bird species relationships with a given environmental variable by continuing the environmental variable line through the origin in the biplot. A perpendicular line was then dropped from each bird species position to the variable of interest. Endpoints of the perpendicular line indicate relative positions of bird species distribution centers along the environmental variable. These endpoints indicate relative relationship of each species to a given variable (ter Braak 1986, 1987).

We used CCA with year as the only environmental axis to plot species scores of the urban-influenced landscape against the rural landscape to document divergence of avian communities. If the avian communities of the 2 landscapes are diverging in opposite directions, a negative relationship should exist. In addition, we used CCA with landscape cover types and structure as covariables and year as environmental variables to measure residual variation. If changes occur over time, some other environmental variables not examined in our study are affecting avian community structure.

RESULTS AND DISCUSSION

Avian Community Change

The trajectories of points over time (centroids of avian community structure) indicate that the avian community in the urban-influenced and rural landscapes diverged along axis 1 and declined along axis 2 (Fig. 1). In addition, the 2 landscapes differ from each other in species composition, even ignoring temporal change, which is not surprising considering differences in land use and vegetation cover types between landscapes. The urban-influenced landscape has more human-intensively managed land, such as cropland, pasture land, and hay meadows, compared to the rural landscape (Boren 1995). The trajectory of both communities progressively diverged over time, but divergence is greater within the rural avian community. Centroid values for the rural avian community between 1966 and 1990 changed by 0.42 and 0.20 SD units for axis 1 and 2, respectively. This suggests avian community structure was strongly affected by aa temporal decrease in deciduous woodlands by prescribed burning and herbicide application to maintain tallgrass prairie in the rural landscape (Boren 1995). Centroid values for the urban-influenced avian community between 1966 and 1990 are only 0.20 and 0.20 SD units apart for axis 1 and 2, respectively. Species scores from CCA, with year as the only variable, of the urban-influenced landscape had a negative relationship with the species scores of the rural landscape. This confirms the DCA results that the avian communities are diverging in opposite directions over time (Fig. 2).

Detrended correspondence analysis provides a scaling of axes in units of compositional turnover (SD units; Hill and Gauch 1980). This scaling provides a robust estimate of beta diversity (Okland et al. 1990) that reflects rate of change in community composition along a gradient (Wilson and Mohler 1983, Noss 1993, Samson and Knopf 1993). Based on the small SD axis units, both avian communities exhibit low beta diversity with relatively small temporal movement along axis 1 (Fig. 1). Therefore, change or turnover in avian community species composition in rural and urban-influenced landscapes appears to be relatively slow between 1966 and 1990.

Although the avian community in urban-influenced and rural landscapes diverged over time, the great-tailed grackle (Quiscalus mexicanus) and rock dove (Columba livia) increased in both landscapes (Table 3). This suggests a temporal increase in generalist species by immigration from nearby source habitats. An aggressive trap and transplant program most likely accounted for the observed increase of wild turkey (Meleagris gallopavo) in both landscapes. We observed none of the 10 species endemic to grasslands (Knopf 1994) in our study area. However, grasshopper sparrow (Ammodramus savannarum) and dickcissel (Spiza americana), secondary species that have exhibited significant declines in grasslands (Knopf 1994), increased in the rural landscape but remained relatively unchanged in the urban-influenced landscape. Grasshopper sparrow and dickcissel declines are localized to areas with inadequate breeding habitats (Knopf 1994). The grasshopper sparrow breeds in fields of several types but prefers vegetation approximately 30 cm tall (Hamel 1992). However, the grasshopper sparrow is sensitive to small changes in its habitat. When herbaceous material becomes too thick or trees encroach on prairies and abandoned fields, these habitats become unsuitable as breeding sites (Bull and Farrand 1988). The dickcissel also requires herbaceous cover (approximately 60 cm tall) for breeding (Hamel 1992). Therefore, prescribed burning and herbivory related to cattle grazing in the rural landscape seem to favor these species by maintaining breeding habitat. The eastern meadowlark (Sturnella magna) and lark sparrow (Calamospiza melanocorys), species of high concern, exhibited relatively little change in both landscapes.

The yellow-breasted chat (Icteria virens), an edge species of high concern that requires dense thickets and brush for nesting habitat (Bull and Farrand 1988), declined in both landscapes. The conversion of deciduous forests to brush-treated lands in the rural landscape and to pasture land and hay meadows in the urban-influenced landscape from 1966 to 1990 (Boren 1995) may account for the decline of this species in both landscapes. The greater prairie chicken (Tympanuchus capido) declined only in the urban-influenced landscape where brush-treated land accounted for only 1% of the total area (Boren 1995). This species nests in habitats of standing residual vegetation from a preceding growing season and is dependent upon stand rejuvenation by fire (Kirsch 1974),

We observed a greater loss of neotropical migrants from the urban-influenced landscape compared to the rural landscape (33% and 3% respectively), which can be attributed to differences in land use and associated management practices. The ratio of neotropical migrants to resident/temperate migrants shifted from 1.2:1 to 0.75:1 in the rural landscape while diverging from 1.2:1 to 0.29:1 in the urban-influenced landscape. Changes in neotropical migrant diversity and density by urban sprawl results from human-induced changes in vegetation composition (Joyce et al. 1990). However, recent scientific studies suggest the primary factors limiting neotropical migrants are related to fragmentation and edge effect as opposed to habitat loss (Hagan and Johnston 1992, Faaborg et al. 1993, Maurer and Heywood 1993, Thompson et al. 1993).

Landscape quality, especially with regard to landscape fragmentation and diversity, continued to erode between 1966 and 1990 in the urban-influenced landscape (Boren 1995), which may account for the observed loss of neotropical migrants from the urban-influenced landscape. Problems associated with habitat fragmentation include increased edge habitat, parasitism rates, predation rates, and isolation effects which generally have adverse effects on neotropical migrant species (Johnson and Temple 1986, Faaborg et al. 1993). Our data also suggests the biological diversity and ecological integrity of the urban-influenced landscape is lower compared to the rural landscape. Neotropical migratory birds provide ideal indices of ecological integrity because they are highly sensitive to changes in landscapes that compromise the spatial continuity and integrity of natural ecosystems (Maurer 1993). However, indices of biological diversity must take into account the dynamic nature of ecosystems and include ecological processes occurring outside the area of interest (Landres 1992).

Differences in avian nesting and foraging zones between landscapes can be attributed to differences in land use and associated management practices. Prescribed burning, herbicide application, and grazing management resulted in a 26% reduction of deciduous woodland in the rural landscape (Boren 1995). Avian community in the rural landscapes shifted from tree nesting species (55% reduction) to ground and shrub nesters which supports our observed reduction of tree foraging to ground foraging species in the rural landscape. However, shifts in nesting and foraging zones are not as apparent in the urban-influenced landscape. In addition, changes in vegetation cover type altered avian community structure by decreasing some forest and edge species in both landscapes relative to prairie and generalist species in the rural and urban-influenced landscapes, respectively (Table 3). Management practices associated with the rural landscape in this study appear to be more conducive to maintaining biodiversity of grassland species. However, community shift towards generalist species in the urban-influenced landscape suggest a continued increase in exotics and species beyond their historical range which pose a significant threat to the loss of native avian assemblages (Knopf 1986, Drake et al. 1989).

Influence of Landscape Cover Type and Structure

Landscape Cover Type.--We expected a strong relationship between land cover types and the distribution of breeding birds (Avery 1989). Indeed, the CCA ordination explained approximately 43% of the variation associated with the relationship between the landscape cover types and both rural and urban-influenced avian data sets (Table 4). The eigenvalues for axes 1 and 2 explained 71 and 78% of the cumulative variance of the bird species-landscape cover type relationship of the rural and urban-influenced data sets, respectively. All land use and vegetation cover types (Table 1) were included in forward selection analysis. Forward selection identified 5 land use and vegetation cover type variables ($\underline{P} < 0.05$) that explained 39% of variation in breeding bird abundances in the rural landscape including forest (17%), cropland (9%), water (5%), developed area (4%), brush-treated land (2%), and roads (2%). Forward selection also identified 5 land use and vegetation cover type variables ($\underline{P} < 0.05$) that explained 38% of variation in breeding bird abundances in the urban-influenced landscape including forest (25%), cropland (4%), roads (4%), water (3%), and native grasslands (2%). Both axes were significant ($\underline{P} < 0.01$) for both landscapes according to Monte Carlo permutation tests.

The relative importance of each environmental variable for predicting the community composition can be found through analysis of canonical coefficients and intraset correlations (ter Braak 1986). Canonical coefficients define the ordination axes as linear combinations of the environmental variables. Intraset correlations are the correlation coefficients between the variables and the axes (ter Braak 1986). Canonical coefficients describe the partial or residual variation and are essentially equivalent to regression coefficients. However, with intraset correlations other variables are assumed to covary with that one environmental variable in the particular way they do in the data set and thus should be used in a multivariate environment. The ordination diagram shows the relationships between the avian community in terms of main axes of variation (Kalkhoven and Opdam 1984).

The variables most correlated with axis 1, based on intraset correlations (Table 5), of the rural landscape were forest and brush-treated land. Thus, axis 1 separates species that decreased and were dependent on deciduous woodland cover (i.e., black and white warbler (Mniotilta varia), pileated woodpecker (Dryocopus pileatus), summer tanager (Piranga rubra), and eastern tufted titmouse (Parus bicolor)) from species that increased and required more open canopy and fewer trees (e.g., barn swallow (Hirundo rustica), dickcissel, and grasshopper sparrow) (Fig. 3).

The variables most correlated with axis 1 of the urban-influenced landscape were forest and native grassland (Table 5). Axis 1 separates species with affinities for forest and shrubland (e.g., chipping sparrow (Spizella passerina), Kentucky warbler (Oporornis formosus), and northern parula warbler (Parula americana)) from species preferring open grasslands (e.g., greater prairie chicken) (Fig. 4). The variables most correlated with axis 2 of the urban-influenced landscape were roads and grassland. Axis 2 separates generalist species that increased and are commonly associated with human development (e.g., American robin (Turdus migratorius), house sparrow (Passer domesticus), purple martin (Progne subis), rock dove, and European starling (Sturnus vulgaris)) from prairie species which declined and are associated with less human disturbance (i.e., greater prairie chicken and cattle egret (Bubulcus ibis)) (Fig. 4).

Different landscape cover types between the landscapes influenced avian community structure in this study. Avian community structure was primarily related to deciduous forest and brush-treated land in the rural landscape compared to deciduous forest, native grassland, and roads in the urban-influenced landscape. This pattern suggests that continued urban sprawl into rural landscapes may result in increased generalist species as the result of increased roads and decreased native grassland. However, inferences on the influence of urban sprawl on rural avifauna must be made with caution. High mobility of birds makes them less dependent on local conditions than sedentary species and avian community structure may be influenced by surrounding bird communities (Jarvinen and Vaisanen 1980).

Landscape Structure.--The CCA ordination explained approximately 18 and 21% of the variation associated with the relationship between the landscape structure and the rural and urban-influenced avian data sets respectively (Table 4). Because landscape cover types explained more than twice the variation of the avian data set compared to the landscape structure variables, landscape cover type ordinations better explain temporal changes in avian community structure in this study. At the landscape scale, avian community composition is a function of vegetation structure (physiognomy) while at the within-stand level particular plant taxonomic composition (floristics) is more important than structure in determining avian community composition (Rotenberry 1985). However, most biodiversity studies have focused on forests or woodland areas, but little research has been conducted in the tallgrass prairie ecosystem. Our results support Roth (1976) and Wiens (1974) comments that generalizations relating vegetation structure and complexity to avian community structure were unrealistic for grasslands. While brush and forests vary broadly in vegetation structure and composition, which correlate with avian diversity, the degree of variability of heterogeneity among grasslands at the landscape scale is so subtle that its affect on avian diversity can be obscured (Knick and Rotenberry 1995) as observed by the inability of our landscape structure variables to explain temporal changes in avian community structure.

MANAGEMENT IMPLICATIONS

Changes in land use and vegetation cover types altered avian community structure in this study. Avian community in the urban-influenced (intensively managed) and rural (extensively managed) landscapes diverged over time because of different land use and management practices associated with each landscape. Temporal shifts in avian community structure are reflected in altered avian biodiversity with increasing prairie and generalist associated species in the rural and urban-influenced landscapes, respectively. Management practices to preserve prairie birds and maintain biological diversity of prairies should encourage increases in the abundance of native plant communities. Maintenance of the tallgrass prairie by prescribed burning, judicious herbicide use for control of exotic plants, and grazing management appear generally conducive to this objective. However, land uses and management practices associated with areas surrounding urban centers pose a threat to preserving the integrity of native plant communities. Although different variables explained avian community structure in the 2 landscapes, our results suggest management practices that alter landscape structure have less impact on community structure than changes in vegetation cover types.

Our data suggest biologists and conservationists should focus more attention on biological diversity of urban and suburban-influenced ecosystems by maintaining native plant communities. In 1989, 74% of the United States population resided in urban areas and that number is expected to increase to >80% by the year 2025 (Haub and Kent 1989). The growth of metropolitan areas in the United States indicates knowledge of ecosystems under the influence of urbanization can only become increasingly important (McDonnell and Pickett 1990). Our results suggest in the absence of societal pressure to halt urban sprawl into rural landscapes, ecosystem integrity will most likely continue to degrade. This further necessitates the need to understand ecological systems along urban-rural gradients to enable biologists to make ecologically sound management of human-dominated ecosystems.

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Table 1. Classification system used to map land use and vegetation cover types (adapted from Stoms et al. 1983).

Land use and cover type	Description
Developed area	Land occupied by residential, industrial, or other human structures and non-agricultural activities. Also includes transportation and utility facilities.
Roads	Black top, gravel, dirt roads and driveways
Water	Ponds, lakes, streams, and rivers
Cropland	Land cultivated for row crops and cereal grains but excluding grazing lands
Pasture land and hay meadows	Includes pasture land (seeded, grasslands used for grazing by cattle, sheep, goats, and horses) and hay meadows
Native grassland	Native grasslands with less than 10% cover by shrubs or trees
Scrub forest	Vegetation dominated (>10%) by cover of broadleaf hardwoods. Mostly post oak (*Quercus stellata*) and blackjack oak (*Q. marilandica*)
Brush-treated land	Native vegetation subjected to herbicides, fire, or chaining to control woody brush encroachment
Bare ground	Land with less than 5% vegetative cover

Table 2. Eigenvalues and cumulative variance (%) of species data for the first 4 axes of detrended correspondence analysis on species data, with year as a passive environmental variable, in a rural (extensively managed) and urban-influenced (intensively managed) landscape.

	Axis 1	Axis 2	Axis 3	Axis 4	Total inertia
Rural					
Eigenvalue	0.30	0.22	0.12	0.09	2.73
Cumulative variance of species data (%)	11.1	18.9	23.5	26.9	
Urban-influenced					
Eigenvalue	0.38	0.18	0.12	0.09	2.91
Cumulative variance of species data (%)	12.9	18.9	23.2	26.4	

Table 3. Avian species responsible for shifts in avian community structure in a rural (extensively managed) and urban-influenced (extensively managed) landscape over a 24-year period, 1966 to 1990. Minor species (those that occurred three or less times) were omitted.

Species	Code	Scientific name	Type[a]	Habitat[b]	Concern[c]	Foraging[d]	Nesting[e]
Rural landscape							
Loss							
Yellow-breasted chat	YBCH	Icteria virens	Neotrop	Edge	High	Ground	Shrub
Blue-gray gnatchatcher	BGGN	Polioptila caerulea	Neotrop	Edge	Moderate	Canopy	Midstory
Greater roadrunner	GRRO	Geococcyx californianu	Resident	Prairie	High	Ground	Shrub
Bewick's wren	BEWR	Thryomanes bewickii	Temp	Edge	High	Ground	Cavity
Black and white warbler	BAWW	Mniotilta varia	Neotrop	Forest	Moderate	Midstory	Ground
Field sparrow	FISP	Spizella pusilla	Temp	Edge	High	Ground	Ground
Painted bunting	PABU	Passerina ciris	Neotrop	Edge	High	Ground	Shrub
Pileated woodpecker	PIWO	Dryocopus pileatus	Resident	Forest	Moderate	Bole	Cavity
Summer tanager	SUTA	Piranga rubra	Neotrop	Forest	High	Midstory	Midstory
Eastern tufted titmouse	ETTI	Parus bicolor	Resident	Forest	High	Midstory	Cavity
White-breasted nuthatch	WBNU	Sitta carolinensis	Resident	Edge	Moderate	Bole	Cavity
Gain							
Dickcissel	DICK	Spiza americana	Neotrop	Prairie	High	Ground	Ground
Wild turkey	WITU	Meleagris gallopavo	Resident	Edge	High	Ground	Ground
Barn swallow	BARS	Hirundo rustica	Neotrop	Develop	Moderate	Aerial	Other
Grasshopper sparrow	GRSP	Ammodramus savannarum	Neotrop	Prairie	High	Ground	Ground
Great-tailed grackle	GTGR	Quiscalus mexicanus	Resident	Edge	Moderate	Ground	Shrub
Little blue heron	LBHE	Egretta caerulea	Temp	Water	Moderate	Water	Shrub
Rock dove	RODO	Columba livia	Resident	Develop	Low	Ground	Other

Table 3. Continued.

Species	Code	Scientific name	Type[a]	Habitat[b]	Concern[c]	Foraging[d]	Nesting[e]
Urban-influenced landscape							
Loss							
Black-billed cucko	BBCU	Coccyzus erythropthaim	Neotrop	Edge	High	Midstory	Shrub
Cattle egret	CAEG	Bubulcus ibis	Resident	Prairie	Low	Ground	Shrub
Yellow-breasted chat	YBCH	Icteria virens	Neotrop	Edge	High	Ground	Shrub
Chipping sparrow	CHSP	Spizella passerina	Neotrop	Forest	Moderate	Ground	Shrub
Common yellowthroat	COYE	Geothlypis trichas	Neotrop	Edge	Moderate	Ground	Shrub
Great-horned Owl	GHOW	Bubo virginianus	Resident	Edge	Moderate	Ground	Cavity
Greater prairie chicken	GPCH	Tympanuchus capido	Resident	Prairie	High	Ground	Ground
Kentucky warbler	KEWA	Oporornis formosus	Neotrop	Forest	High	Ground	Ground
Northern-parula warbler	NOPA	Parula americana	Neotrop	Forest	High	Midstory	Canopy
Red-shouldered hawk	RSHA	Buteo lineatus	Temp	Edge	Moderate	Ground	Canopy
Yellow-bellied sapsucker	YBSA	Sphyrapicus sp.	Temp	Edge	High	Bole	Cavity
Gain							
American robin	AMRO	Turdus migratorius	Temp	Develop	Low	Ground	Shrub
Gray catbird	GRCA	Dumetella carolinensis	Neotrop	Edge	High	Ground	Shrub
Common grackle	COGR	Quiscalus quiscula	Resident	Edge	Low	Ground	Midstory
Great-tailed grackle	GTGR	Quiscalus mexicanus	Resident	Edge	Moderate	Ground	Shrub
House sparrow	HOSP	Passer domesticus	Resident	Develop	Low	Ground	Cavity
Purple martin	PUMA	Progne subis	Neotrop	Develop	Moderate	Aerial	Cavity
Rock dove	RODO	Columba livia	Resident	Develop	Low	Ground	Other
European starling	EUST	Sturnus vulgaris	Resident	Develop	Low	Ground	Cavity

Table 3. Continued.

Species	Code	Scientific name	Type[a]	Habitat[b]	Concern[c]	Foraging[d]	Nesting[e]
Wild turkey	WITU	Meleagris gallopavo	Resident	Edge	High	Ground	Ground

[a]Species classified as neotropical migrants (Neotrop), temperate migrants (Temp), and residents (Resident).

[b]Species grouped into designations of habitat occurrence: forest (Forest), forest edge and shrubland (Edge), prairie (Prairie), and developed areas (Developed).

[c]Species grouped into population trends: low concern (Low), moderate concern (Moderate), and high concern (High).

[d]Species grouped into foraging zones: open zones (Aerial), foliage 0 - 3 m (Ground), foliage 3 - 10 m (Midstory), and trunks and limbs (Bole).

[e]Species grouped into nesting zones: ground (Ground), 0 - 3 m (Shrub), 3 - 10 m (Midstory), > 10 m (Canopy), cavity (Cavity), and variable heights and substrates (Other).

Table 4. Eigenvalues, correlation coefficients, and cumulative variances (%) between species and environmental axes for stepwise canonical correspondence analyses carried out on landscape cover type and landscape structure variables in a rural (extensively managed) and urban-influenced (intensively managed) landscape.

	Landscape cover type		Landscape structure	
	Axis 1	Axis 2	Axis 1	Axis 2
Rural				
Eigenvalue[a]	0.18	0.12	0.08	0.05
Species-environment correlation[b]	0.80	0.74	0.55	0.55
Cumulative variance explained (%)[c]	43.1	70.6	43.4	68.8
Sum of all canonical eigenvalues[d]	0.43		0.18	
Total inertia	2.74		2.74	
Urban-influenced				
Eigenvalue[a]	0.28	0.05	0.09	0.06
Species-environment correlation[b]	0.88	0.60	0.59	0.52
Cumulative variance explained (%)[c]	65.1	77.6	41.5	68.2
Sum of all canonical eigenvalues[d]	0.44		0.21	
Total inertia	2.91		2.91	

[a]Eigenvalues (λ) measure the importance of the ordination axis.

[b]Species-environment correlation (r) is a measure of how well the extracted variation in community composition can be explained by the environmental variables.

[c]Cumulative percentage variance of species-environment relation.

[d]Sum of all canonical eigenvalues represents the total amount of extracted variation accounted for by the CCA ordination.

Table 5. Canonical coefficients and intraset correlations for variables of the stepwise canonical correspondence analysis carried out on landscape cover type and structure in a rural (extensively managed) and urban-influenced (intensively managed) landscape.

	Canonical coefficients		Intraset correlations	
	Axis 1	Axis 2	Axis 1	Axis 2
Rural				
Landscape cover types				
Developed area	0.1886	0.3512	0.3773	0.2696
Cropland	0.2381	0.5417	0.4600	0.5967
Pasture land/hay meadows	0.0137	0.0671	0.2043	0.3519
Native grassland	-0.3543	0.1984	-0.4498	0.3153
Scrub forest	0.4993	-0.4171	0.8592	-0.4816
Brush-treated land	-0.3345	0.0373	-0.4836	-0.1067
Roads	-0.0772	0.1264	-0.2431	0.2208
Water	0.1993	0.4134	0.2424	0.6642
Bare ground	-0.0204	0.0211	0.0368	-0.1659
Landscape structure				
Mean patch size	0.2962	-0.1888	0.6066	-0.3657
Fractal dimension	0.3953	0.3719	0.1616	0.5995
Richness	-0.6962	3.8510	-0.6140	0.5035
Shannon diversity	-0.9746	-4.4340	-0.5840	0.2190
Dominance	-1.2727	-4.0655	0.2108	0.0841
Contagion	1.3820	-0.4068	0.2636	0.4771
Angular second moment	-0.4582	1.0816	0.7338	0.2551

Table 5. Continued.

	Canonical coefficients		Intraset correlations	
	Axis 1	Axis 2	Axis 1	Axis 2
Urban-influenced				
Landscape cover types				
Developed area	-0.0909	0.2804	-0.1122	0.4117
Cropland	-0.1380	0.2842	0.2237	0.2307
Pasture land/hay meadows	-0.4134	0.4857	-0.2183	0.4710
Native grassland	-0.6146	-0.0212	-0.4644	-0.5497
Scrub forest	0.5798	0.2718	0.9362	0.0521
Brush-treated land	0.0144	-0.0606	0.1327	-0.1508
Roads	-0.0822	0.6983	-0.2805	0.7025
Water	0.2035	0.0395	0.3795	-0.2206
Bare ground	-0.0030	-0.3156	0.0046	-0.2329
Landscape structure				
Mean patch size	0.5554	0.4059	0.8089	0.3836
Fractal dimension	-0.0703	-0.3182	-0.3795	-0.2690
Richness	-0.4051	2.1382	-0.4256	-0.4442
Shannon diversity	-0.1789	-0.6500	-0.7334	-0.1809
Dominance	0.0057	0.1669	0.6437	-0.1278
Contagion	0.8441	-2.8708	0.6586	-0.4575
Angular second moment	-0.4587	2.3910	0.7392	0.1250

Fig. 1. Detrended correspondence analysis (DCA) ordination of centroids for avian community structure on the rural (extensively managed) and urban-influenced (intensively managed) landscapes. Lines indicate trajectories of avian community change between 1966 and 1990 defined by the ordination axes.

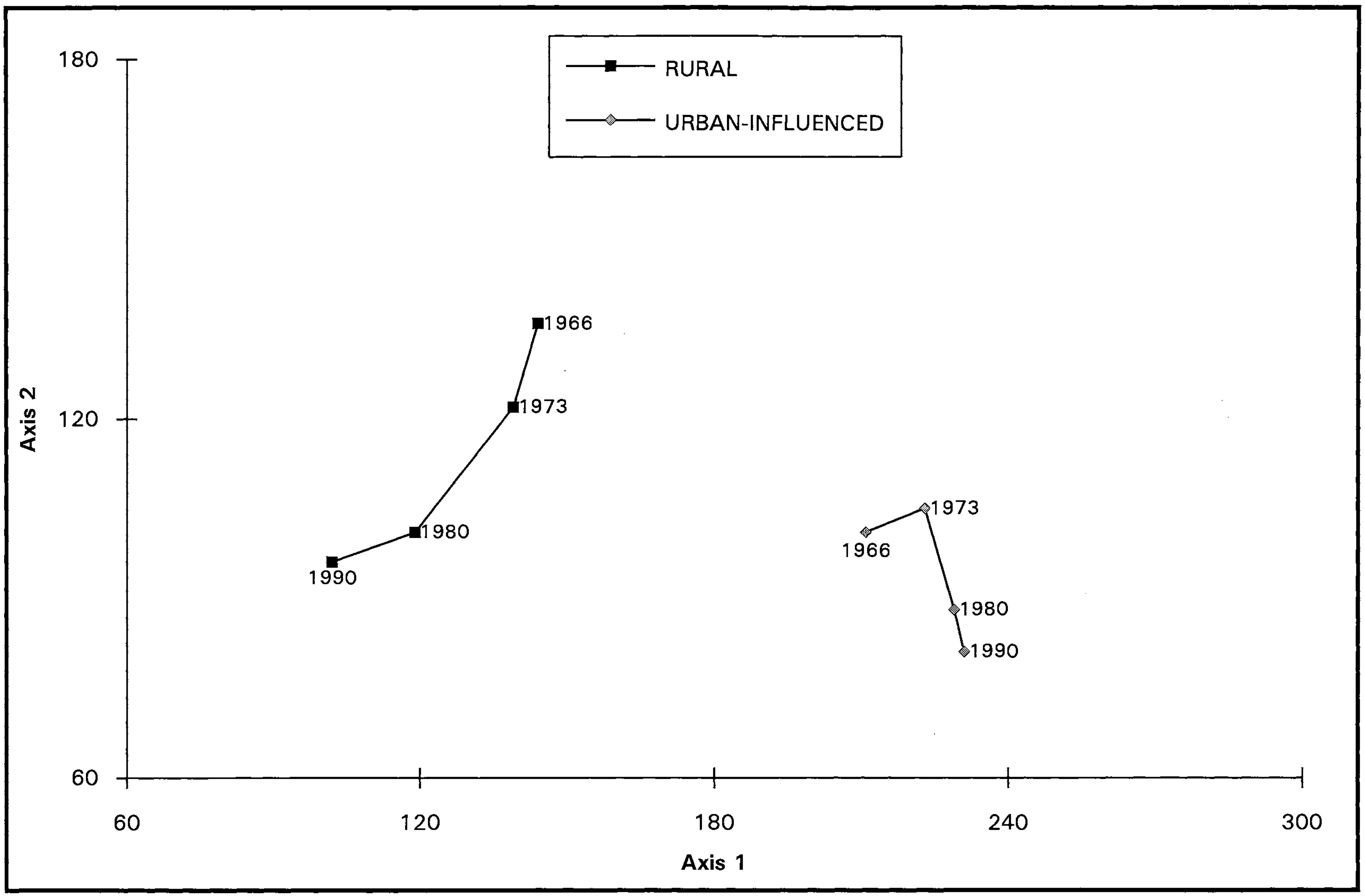

RURAL
URBAN-INFLUENCED
180
120
60
Axis 2
60
120
180
240
300
Axis 1
1966
1973
1980
1990
1973
1966
1980
1990

Fig. 2. Species scores from canonical correspondence analysis (CCA), with year as the only variable, of the urban-influenced landscape against the plotted species scores of the rural landscape ($r^2 = 0.13$, $P < 0.05$).

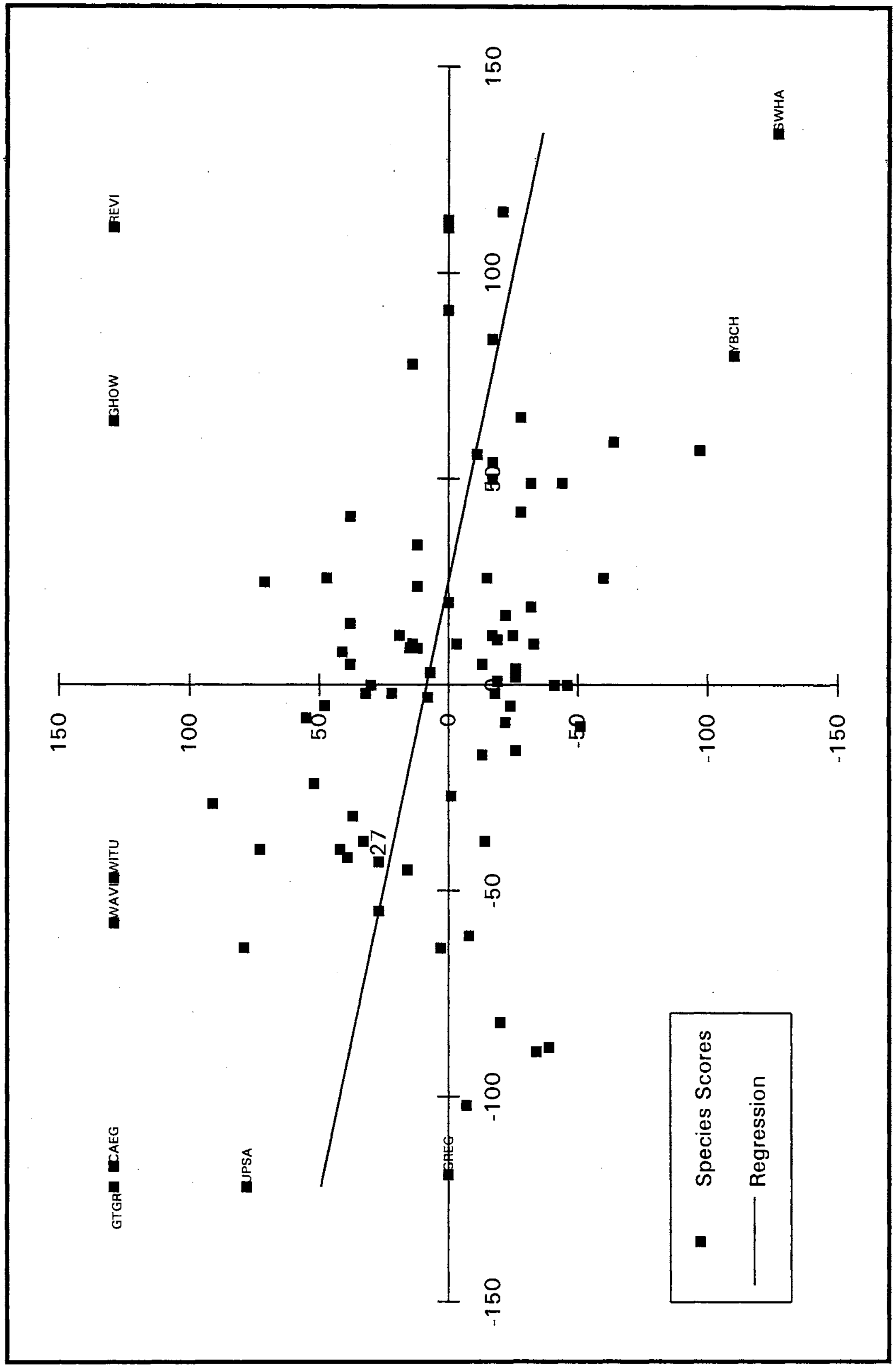
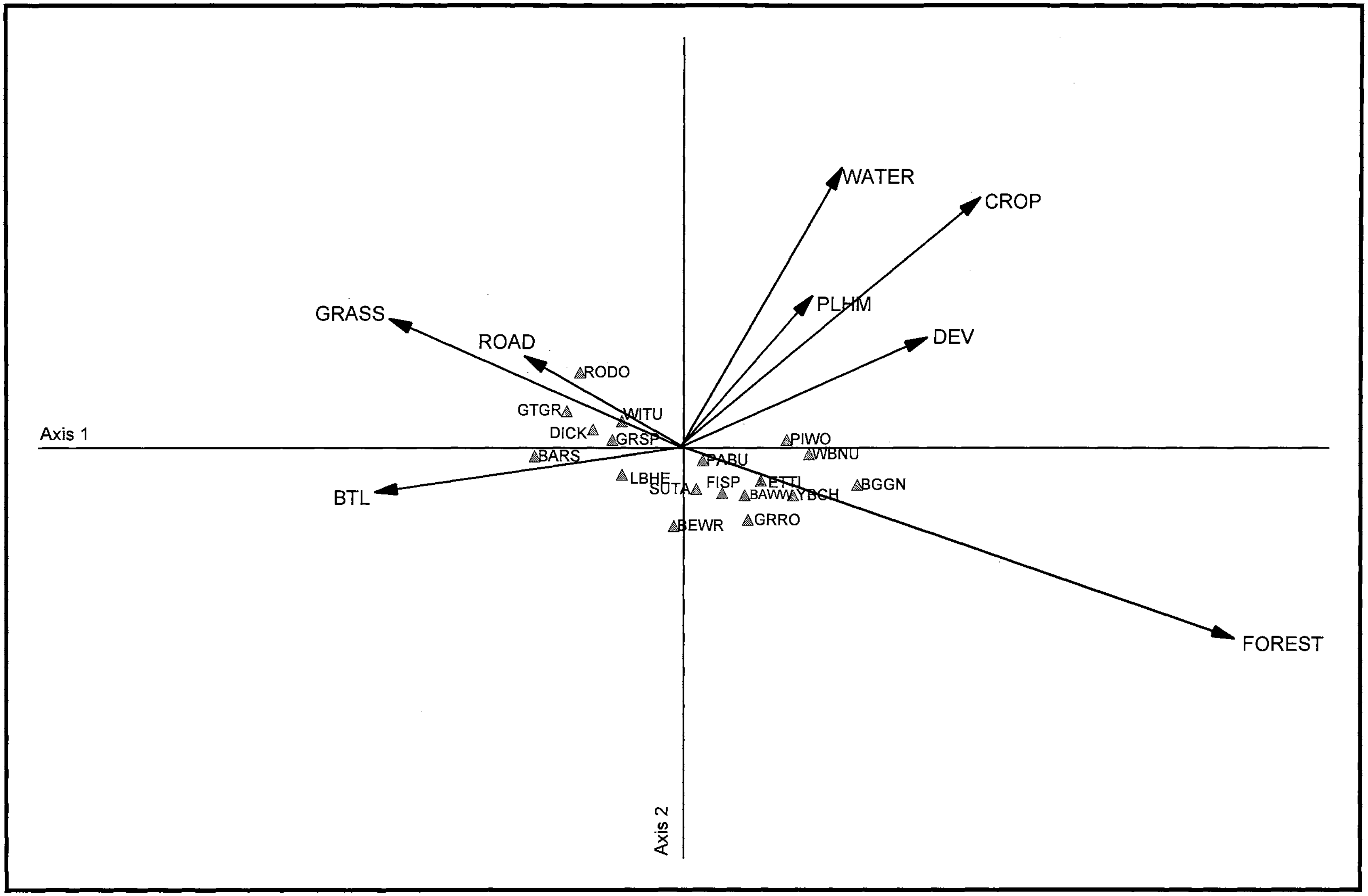
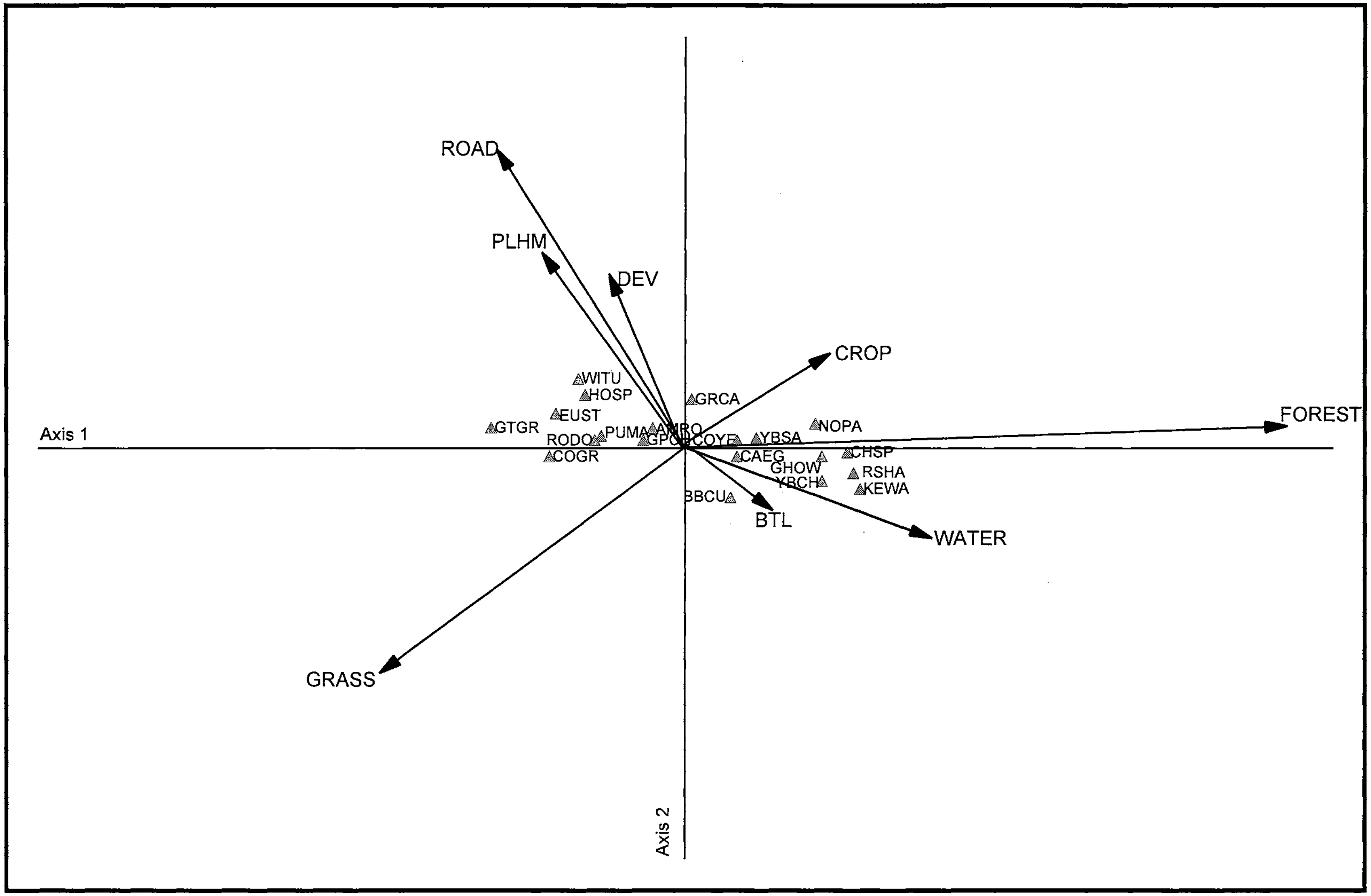

Species Scores
Regression
REVI
GHOW
WITU
WAVI
CAEG
GTGR
UPSA
GREG
YBCH
SWHA
27
150
100
50
0
-50
-100
-150

Fig. 3. Distribution of 18 species of birds in the rural (extensively managed) landscape. Canonical correspondence analysis (CCA) ordination diagram with birds (▲) and environmental variables (vegetation cover types; arrows). The bird species are: YBCH = yellow-breasted chat, BGGN = blue-gray gnatchatcher, GRRO = greater roadrunner, BEWR = bewick's wren, BAWW = black and white warbler, FISP = field sparrow, PABU = painted bunting, PIWO = pileated woodpecker, SUTA = summer tanager, ETTI = eastern tufted titmouse, WBNU = white-breasted nuthatch, DICK = dickcissel, WITU = wild turkey, BARS = barn swallow, GRSP = grasshopper sparrow, GTGR = great-tailed grackle, LBHE = little blue heron, and RODO = rock dove. Environmental variables are: DEV = developed area, ROAD = road, WATER = water, CROP = cropland, PLHM = pasture land and hay meadows, GRASS = native grassland, FOREST = scrub forest, and BTL = brush-treat land.



WATER
CROP
PLHM
DEV
GRASS
ROAD
RODO
GTGR
WITU
DICK
Axis 1
GRSP
BARS
PIWO
WBNU
PABU
LBHE
SUTA
FISP
ETTI
BAWW
YBCH
BGGN
BTL
BEWR
GRRO
FOREST
Axis 2

Fig. 4. Distribution of 20 species of birds in the urban-influenced (intensively managed) landscape. Canonical correspondence analysis (CCA) ordination diagram with birds (▲) and environmental variables (vegetation cover types; arrows). The bird species are: BBCU = black-billed cucko, CAEG = cattle egret, YBCH = yellow-breasted chat, CHSP = chipping sparrow, COYE = common yellowthroat, GHOW = great-horned owl, GPCH = greater prairie chicken, KEWA = Kentucky warbler, NOPA = northern-parula warbler, RSHA = red-shouldered hawk, YBSA = yellow-bellied sapsucker, AMRO = American robin, GRCA = gray catbird, COGR = common grackle, GTGR = great-tailed grackle, HOSP = house sparrow, PUMA = purple martin, RODO = rock dove, EUST = European starling, and WITU = wild turkey. Environmental variables are: DEV = developed area, ROAD = road, WATER = water, CROP = cropland, PLHM = pasture land and hay meadows, GRASS = native grassland, FOREST = scrub forest, and BTL = brush-treated land.


ROAD
PLHM
DEV
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Axis 1
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WITU
HOSP
GRCA
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GTGR
NOPA
PUMA
AMRO
RODO
COYE
YBSA
CHSP
COGR
CAEG
GHOW
RSHA
YBCH
KEWA
BBCU

CHAPTER IV

VEGETATION COVER TYPE AND AVIAN SPECIES CHANGES ON LANDSCAPES WITHIN A WILDLAND-URBAN INTERFACE

Summary

1. Probability of occurrence of selected avian species were modeled as a function of modeled changes in landscape cover types in two landscapes to test whether 1) exotic and generalist avian species will continue to increase in a urban-influenced landscape; and, 2) native grassland avian species will continue to increase in a rural landscape.

2. Landscape cover types were modeled with logistic regression based on temporal changes between 1966 and 1990. Demographic-economic regression models also were used to predict landscape cover types in year 2014 based on selected independent variables.

3. Logistic regressions were used to model the probability of occurrence of selected avian species based on predicted area of landscape cover types in year 2014.

4. Model output of vegetation cover types suggests the continued use of intensive management practices in the urban-influenced landscape while extensive management practices will maintain the native vegetation component in the rural landscape.

5. Our models suggest continued intensive agriculture practices associated with urban-influenced landscapes will adversely affect native grassland bird species to a greater magnitude than extensive ranching practices in the rural landscapes.

Introduction

Throughout most of the world human activities have become the primary influence on ecosystems. In 1989, 74% of the United States population resided in urban areas and that number is expected to increase to >80% by the year 2025 (Haub & Kent 1989). This growth of metropolitan areas in the United States comes at the expense of continued urban sprawl into rural landscapes (McDonnell & Pickett 1990). Land use changes associated with urban sprawl often reduce ecosystem diversity on a regional scale because of the replacement of natural vegetation with intensively managed systems of altered structure and composition (Davis & Glick 1978; Krummel et al. 1987; McNeely et al. 1990). Increased agriculture intensification into rural landscapes of the eastern Great Plains results in altered land ownership patterns and management practices resulting in contrasting vegetation cover types between rural and urban-influenced landscapes (Boren 1995). Intensive management practices associated with the urban influenced landscapes results in cropland, introduced pasture land, and hay meadows as the dominant cover types. However, extensive management practices associated with rural landscapes results in native vegetation as the dominant cover type.

In North America's native grasslands, which are used mostly in extensive agriculture by grazing livestock, native endemic species have declined in numbers while exotic species have increased simultaneously (Askins 1993; Knopf 1994). In fact, native birds in North America's grasslands have undergone more widespread declines over the past 25 years than any other U.S. bird group, which warrants the increasing concern for the conservation of these birds (Knopf 1994). Our results from a study conducted in a tallgrass prairie-deciduous forest ecotone indicate intensive agriculture practices preceding urban sprawl are associated with increasing exotic and generalist species at the expense of endemic species (Boren 1995).

Conservation of native grassland species may be improved by predictions of future land cover type changes and associated avian composition. Predictions of future landscape cover types are an important component to understanding the ecological dynamics necessary to integrate the often conflicting demands of wildlife habitat, agriculture, and urban development. Multivariate analysis techniques in combination with logistic regression can provide important insight to landscape-bird community relationships (Braithwaite et al. 1989; Eyre et al. 1992; Osborne & Tigar 1992). Predictive statistical models for avian composition under different landscape management scenarios also will better enable biologists to assess the effects of continued land use change on avian diversity.

Our objective was to model the probability of occurrence of selected avian species as a function of modeled changes in landscape cover types in two landscapes with contrasting anthropogenic influences. Our premise was that intensified management practices in an urban-influenced landscape will continue to result in increased intensively managed landscapes at the expense of extensively managed natural landscapes. In contrast, continued extensive management practices associated with a rural landscape will result in increased landscape coverage by native vegetation. Based on this premise, we hypothesized that 1) exotic and generalist avian species will continue to increase in the urban-influenced landscape; and, 2) native grassland avian species will continue to increase in the rural landscape.

Study area

Our study was centered around suburban Tulsa, Oklahoma, and included surrounding wildlands in northeastern Osage and southern Washington counties. The selection of the study area was based on areas with a suburban-wildland transition and areas in which biological diversity of bird species, including species richness and evenness, were available. The study area included two U.S. Fish and Wildlife Service Breeding Bird Survey (BBS) routes: 024 (Collinsville) and 026 (Bartlesville). Legal description of these survey routes are provided by Baumgartner & Baumgartner (1992).

The BBS routes lie on an ecotonal area between the Cherokee Prairie grassland formation and oak-hickory savanna of the Cross Timbers (Bruner 1931; Soil Conservation Service 1981). The Cherokee Prairie of Oklahoma extends as a long, narrow strip, 240 km southward from the Kansas state line with a width ranging from 48 to 96 km throughout most of its length. The area is better adapted to support grasses, forbs, and legumes than forests because of climate and underlying geology (Harlan 1957). The Cross Timbers of Oklahoma lie west of the Cherokee Prairie and the Lower Arkansas Valley, extending 288 km southward from Kansas and approximately 80 km wide. The region is a transitional oak forest with interspersed prairie (Bruner 1931; Gray & Galloway 1959).

Survey routes also varied in their proximity from Tulsa, a major metropolitan area in northern Oklahoma with a estimated population of 361,628 (U.S. Department of Commerce 1990). The Collinsville route is 24 km from Tulsa, in Washington County and the Bartlesville route is 74 km from Tulsa, in Osage County. A 50% increase in human use areas was observed in the Collinsville route while a 4% decrease was observed in the Bartlesville route between 1966 and 1990 (Boren 1995). Human population density of Washington and Osage County in 1990 was 3340 km^{-2} and 520 km^{-2}, respectively. In addition, land ownership size was typically smaller in the Collinsville route compared to the Bartlesville route. We considered the Collinsville route to be subjected to more urban influence than the Bartlesville route. Thus, we viewed the Collinsville route as an intensively managed landscape and Bartlesville route as an extensively managed landscape, and from this point forward, the two landscapes will be discussed as urban-influenced or rural.

Model construction

BIRD SURVEYS AND DATABASE

Our models were derived from bird abundance data from BBS routes conducted by the U.S. Fish and Wildlife Service. Each 40·2 km route is conducted on secondary roads and consists of 50 BBS stops 800 m apart (Bystrak 1981). The BBS data set is the only data set that indexes the population status of many species of birds over a large geographical area and time (Bystrak 1981; Geissler and Noon 1981). Although a roadside count misses some species and is limited by placement of roads, the results are believed to be a fairly reliable index for a prairie-woodland ecosystem (Baumgartner & Baumgartner 1992).

Land use and vegetation cover types for 1966, 1973, 1980, and 1990 on both landscapes were examined using a Geographic Information System (GIS) (Boren 1995). Landscape cover types identified on each landscape included developed areas, roads, water, cropland, pasture land and hay meadows, native grassland, deciduous forest, brush-treated land, and bare ground (Table 1).

Bird abundances, available from 1967 to 1991, were lumped around 4 years for which landscape cover type data were documented previously for both landscapes for detrended and canonical correspondence analysis (Boren 1995). Thus, breeding bird data from 1967 to 1970 corresponded to the 1966 landscape data, BBS data from 1971 to 1976 corresponded to the 1973 landscape data, BBS data from 1977 to 1984 corresponded to the 1980 landscape data, and BBS data from 1985 to 1991 corresponded to the 1990 landscape data. Relative abundance was then calculated for each of the four time periods by averaging relative abundance for the 4 years.

AVIAN COMMUNITY STRUCTURE ANALYSIS

Detrended correspondence analysis (DCA) was performed on the BBS data with CANOCO (ter Braak 1988) to determine if avian community structure differed between landscapes and to document shifts in avian community structure over time by using year as the passive environmental variable. We used species scores generated by DCA to determine the avian species responsible for temporal shifts in avian community structure. Therefore, DCA provided a multivariate approach identifying the species declining or increasing within each landscape which were modeled as a function of changes in landscape cover types.

Canonical correspondence analysis (CCA) was performed with CANOCO (ter Braak 1988) to determine the influence of landscape cover type and landscape structure on the breeding bird community for each landscape. By applying CCA it is possible to identify the important environmental variables that explained community structure with no a priori knowledge about the possible predictor variables (Saetersdal & Birks 1993). Further discussion on the DCA and CCA analysis is provided in Boren (1995).

LANDSCAPE COVER TYPE MODELS

Landscape cover types were modeled based on temporal changes between 1966 and 1990. Logistic regression (PROC LOGISTIC; SAS 1988) was used for predicting probability of landscape cover type occurrence on each BBS stop in the rural and urban-influenced landscapes with year as the independent variable. Projected area of landscape cover types in year 2014 for each BBS stop was determined by multiplying the area of a BBS stop (50·2 ha) by the probability of occurrence. The models assume temporal changes in landscape cover types between 1966 and 1990 continue at the same rate into the future.

Demographic-economic regression models also were used to predict the area (ha) of landscape cover types in year 2014 for both landscapes. These multiple regression models (PROC REG; SAS 1988) were based on selected independent variables predicted for year 2014. Independent variables included rural population density (number people/km^2), 7-year cumulative oil price (U.S. $/barrel), herbicide price ($/ha), 5-year cumulative cattle price margin (difference between 202·5 kg buy and 303·75 kg sell price of stocker cattle), average farm size (ha), and number of farms per county. Only the models that were significant ($\underline{P} < 0{\cdot}05$) were included in the analysis to predict the area (ha) of landscape cover types in year 2014. Area for each projected landscape cover type was adjusted to a per BBS stop basis.

Each independent variable was predicted for year 2014 based on the literature and univariate regression. Predicted rural population density in year 2014 was 5·64 and 15·49 people/km^2 in Osage County (rural landscape) and Washington County (urban-influenced landscape), respectively (Selland & Shahidullah 1988; Oklahoma Population Reports 1981). Rural was defined as having less than 2,500 people (U.S. Department of Commerce 1994). Seven-year cumulative oil price in year 2014 was based on two scenarios (Hawdon 1989). Scenario 1 assumes oil prices begin to rise when most of the excess productive capacity outside the Gulf producing region is eliminated. Until then, prices are assumed to be under constant downward pressure because of the existence of significant excess oil production capacity. Scenario 2 assumes Gulf producers are unable to raise above prices in the 1990's or that market conditions are not conducive to real price increases. Technological innovations supporting scenario 2, such as new conventional and non-conventional (synthetic) supplies of oil, are available that would mitigate against a price increase. Seven-year cumulative oil price was predicted at 236·25 and 127·05 U.S. \$/barrel for scenario 1 and 2, respectively. Predicted herbicide price in year 2014 was 284·38 U.S. \$/ha based on regression ($\underline{R}^2$ = 0·96, $\underline{P}$ = 0·024) from past prices (J.F. Stritzke, personal communication; Hammond et al. 1975; Stritzke 1981). Five-year cumulative cattle price margin was predicted at 751·57 U.S. \$/head based on regression ($\underline{R}^2$ = 0·68, $\underline{P}$ = 0·005) from past price margins (C.E. Ward, personal communication). Predicted average farm size in Washington County was 120 ha based on regression ($\underline{R}^2$ = 0·58, $\underline{P}$ = 0·049) from past farm sizes (U.S. Department of Commerce 1966, 1977, 1984, 1994). Number of farms in Washington County was predicted at 640 farms based on regression ($\underline{R}^2$ = 0·83, $\underline{P}$ = 0·042) from past farm numbers (U.S. Department of Commerce 1966, 1977, 1984, 1994). A farm was defined as places that sold at least 1,000 U.S. dollars of agricultural products per year (U.S. Department of Commerce 1994). By definition, farms included agriculture land used for crops, introduced pasture, and livestock grazing.

AVIAN SPECIES OCCURRENCE MODELS

Bird species determined by DCA to be responsible for shifts in avian community structure between 1966 and 1990 were related to the area (ha) of landscape cover types by logistic regression (PROC LOGISTIC; SAS 1988). Logistic regression has the advantage of assuming that a species' occurrence relates to an environmental gradient in a logistic rather than a linear manner (Osborne & Tigar 1992). Logistic regression models of the form $\log_e[(p/1-p)]=b_0+b_1x_1+b_2x_2$ where x_n is a independent (predictor) variable and b_0 and b_n are parameters or regression coefficients (ter Braak & Looman 1987), provide a means to predict probability of occurrence of species in relation to environmental variables (Heliovaara et al. 1991; Osborne & Tigar 1992). Presence and absence of the bird species were used as the dependent (response) variables. The independent variables, area of each landscape cover type, were tested ($P < 0{\cdot}05$) as linear, quadratic, cubic, and quartic.

We used logistic regressions to model the probability of occurrence of each bird species for each landscape separately based upon the area of each landscape cover type (Eyre et al. 1992; Osborne & Tiger 1992). Predicted area of landscape cover types in year 2014 was used as the independent variable to determine the probability of occurrence for each species on each BBS stop. The probability of occurrence of each bird species was determined by averaging the probability of occurrence of the 50 BBS stop locations for each landscape. Landscape cover types that could not be modeled were assumed to be the same area (ha) in 2014 as in 1990. Frequency of occurrence for each species in the 1990 BBS data for each landscape was determined for comparative purposes.

Model output

LANDSCAPE COVER TYPES

Native grassland, deciduous woodland, brush-treated land, and cropland were modeled in the rural landscape; while native grassland, cropland, and pasture land and hay meadows were modeled in the urban-influenced landscape based on temporal changes in vegetation cover types between 1966 and 1990 (Table 2). Models for other landscape cover types were not statistically significant. Model output of vegetation cover types suggests a continued use of intensive management practices in the urban-influenced landscape while extensive management practices will maintain the native vegetation component in the rural landscape. Based on our model, cropland will decrease about 20% in both the rural and urban-influenced landscape between 1990 and 2014. Reduction in cropland may result from continued decrease in cultivation of marginal lands (Sampson & Knopf 1994). However, pasture land and hay meadows will increase 17% in the urban-influenced landscape over the same time period (Table 3), suggesting continued increase in intensive management practices.

Native grasslands were the dominant cover type for both landscapes in 1990 and 2014 (Table 3). Based on our model, native grasslands will decrease 7% and 10% in the urban-influenced and rural landscapes between 1990 and 2014, respectively. However, bush treated land will increase 24% while relatively little change in deciduous woodlands will occur over the same time period in the rural landscape. This model suggests brush treatment efforts are for the maintenance of existing native grasslands rather than converting existing deciduous woodlands to grasslands. Maintenance of tallgrass prairie dominance in this region requires fire or herbicides to prevent encroachment of woody species (Bragg & Hulbert 1976; Knight, Briggs & Nellis 1994). Therefore, increasing use of extensive management practices to maintain the tallgrass prairie may account for the observed decline in native grasslands along the rural landscape.

A variety of demographic-economic variables modeled influenced the vegetation composition of the landscapes (Table 3). Rural population density was predicted to increase 13% and 37% between 1990 and 2014 in the rural and urban-influenced landscapes, respectively (Oklahoma Population Reports 1981; Selland & Shahidullah 1988). We investigate the effects of rural population density on landscape composition based on the premise that increased population density would result in increased deciduous woodland and decreased native grassland cover. These changes in cover types would result in part because of decreased brush treatment efforts associated with smaller management unit sizes in the rural landscape and the desire for woodland lots for residential development in the urban-influenced landscape. Increasing preference for housing developments with many mature trees over developments with few trees has been observed at the rural-urban fringe (Sullivan 1994). Our model supported the above premise that in a urban-influenced landscape, increased population density will result in increased woodland cover because of the desire for wooded residential lots (Table 4). The predicted rural population density in the urban-influenced landscape was associated with a 22% decrease in native grassland and a 140% increase in deciduous woodlands between 1990 and 2014 (Table 3). However, a negative relationship between population density and both native grassland and deciduous woodland cover was observed with this model in the rural landscape (Table 4). This relationship is most likely because of increased brush treatment practices (52% from 1990 to 2014) which would decrease both native grassland and woodland cover. This suggests other economical or societal pressures for brush treatments may result as the rural population density in the rural landscape increases. For example, increased woodland cover associated with urban sprawl may result in increased pressure for ranching operations to maintain native grasslands for livestock production.

We investigated the effects of number of farms and average farm size on landscape composition in the urban-influenced landscape (Table 4). Number of farms and average farm size was predicted to decrease 2% and 10% between 1992 and 2014, respectively. Our premise was that as farm numbers and average farm size declined in an intensively managed landscape, woodland and native grassland cover would increase because less proportion of the landscape would be managed intensively for cropland, pasture land, and hay meadows. Intensive agriculture practices associated with farms located near urban areas typically reduce native vegetation cover in the Great Plains (Knopf 1994). We also suspected that as average farm size decreased, less brush treatment would be used resulting in a increase in woodland cover. Brush treatment practices, including prescribed burning, become difficult to implement when treatment areas are small or roads and residential areas are nearby (Bidwell & Masters 1994). Results from our model support these premises. The predicted number of farms in the urban-influenced landscape resulted in a 11% and 4% increase in native grassland and deciduous woodland cover between 1990 and 2014, respectively (Table 3). In addition, the predicted average farm size resulted in a 8% and 22% increase in both cover types over the same time period.

We also investigated the effects of economics, including herbicide price, oil price, and cattle price margin, on landscape composition in the rural landscape (Table 4). A 75% increase in herbicide price was predicted between 1990 and 2014. Because of the increased economic burden for a landowner to control woody plant encroachment by chemical means, we suspected that as herbicide price increases native grassland cover would decrease and woodland cover would increase. Although our model predicts native grassland cover to decrease as herbicide price increases, the demographic-economic regression model demonstrated a positive relationship between herbicide price and brush-treated land (Table 4). The predicted herbicide price resulted in a 79% increase in brush-treated land and a 24% decrease in both native grassland and woodland cover. A increase in spatial and temporal application of fire for brush treatment will be necessary as herbicides increase in price.

We modeled 7-year cumulative cattle price margin (Table 4) in the rural landscape under the premise that increased price margin would provide an economic incentive for the land owner to apply brush treatments to increase native grassland cover for livestock production. Cattle price margin was predicted to increase 46% between 1994 and 2014. The model predicts a 39% increase in brush-treated lands and a 5% decrease in woodland cover between 1990 and 2014 (Table 3). Native grasslands will decline by 24% over the same time period. However, the decline in native grasslands may be misleading because grasslands subjected to either prescribed burning or herbicide application are modeled as brush-treated land but they are grasslands by physiognomy.

Agriculture and oil are critically important to the economy of Oklahoma (Woods, Nelson & Bliss 1989). In fact, agriculture and oil combined were responsible for over 56% of the state's employment in 1989. Therefore, increased oil prices may stimulate the rural economy resulting in increased capital for the local rancher to apply brush treatment practices and to purchase more livestock. Therefore, we modeled 5-year cumulative oil price (Table 4) in the rural landscape under the premise that as oil prices increase native grassland cover would increase and deciduous woodland cover would decrease. Based on Hawdon (1989), we predicted the 5-year cumulative oil price to increase 57% with scenario 1 (assumes oil prices rise when excess productive capacity outside the Gulf producing region is eliminated) and decrease 17% with scenario 2 (assumes Gulf producers are unable to raise oil prices) between 1990 and 2014. Scenario 1 resulted in a 27% increase in brush-treated lands between 1990 and 2014 (Table 3). Although the demographic-economic regression model demonstrated little change in woodland cover, native grassland cover declined 12% over the same time period, which may again be misleading because grasslands subjected to brush control are modeled as brush-treated lands. Predicted oil price based on scenario 2 resulted in a 13% reduction in brush-treated land between 1990 and 2014. Although a 9% increase in native grassland cover was observed, woodland cover increased 19% over the same time period.

Our best model, based on $\underline{R}^2$, for predicting native grasslands, deciduous woodland, and brush-treated lands in the rural landscape is based on the variables cattle price margin and oil price (Table 4). The predicted increase in both cattle price margin and oil price (scenario 1) resulted in a 50% increase in brush-treated land and 9% decrease in woodland cover between 1990 and 2014 (Table 3). The predicted increase in cattle price margin and decrease in oil price (scenario 2) also resulted in a increase in brush-treated land and a decrease in woodland cover. However, brush-treated land increased only 9% while woodland cover decreased only 7% between 1990 and 2014. Although oil price appears to mitigate the effects of cattle price margin, our results suggests cattle price margin has a greater influence on extensive management practices that affect brush-treated land and woodland cover in the rural landscape.

In comparison, our best model, based on $\underline{R}^2$, for predicting the same cover types in the urban-influenced landscape is based on the variables average farm size and oil price (Table 4). When modeled with the predicted decrease in average farm size and increase in oil price (scenario 1), we observed relatively little change in woodland cover between 1990 and 2014. However, native grassland cover increased 13% over the same time period (Table 3). This suggests increased oil prices may mitigate the effects of decreased farm size, which typically increase woodland cover at the expense of native grasslands, in the urban-influenced landscape. However, when modeled with the predicted decrease in both average farm size and oil price (scenario 2), woodland cover increased 41% while native grassland cover increased only 3% between 1990 and 2014.

THE AVIAN COMMUNITY

Avian community in the urban-influenced and rural landscapes diverged from one another between 1966 and 1990 because of divergence in land use and management practices associated with each landscape (Boren 1995). Temporal shifts in avian community structure is reflected in altered avian diversity with increasing prairie and generalist associated species in the rural and urban-influenced landscapes, receptively (Table 5; Boren 1995). Avian community structure is primarily related to deciduous forest and brush-treated land in the rural landscape compared to deciduous forest, native grassland, and roads in the urban-influenced landscape (Boren 1995). We modeled bird species responsible for shifts in avian community structure between 1966 and 1990 to predict the probability of occurrence in year 2014 to assess future landscape cover types on the aviafauna (Table 6).

In general, we observed little difference between the probability of species occurrence modeled by logistic regression and demographic-economic regression in the rural landscape (Table 7) because there was little difference in the area of native grassland, deciduous woodland, or brush-treated land between these models (Table 3). We also observed little change in the probability of occurrence, regardless of the model used, of the barn swallow (Hirundo rustica), bewick's wren (Thryomanes bewickii), painted bunting (Passerina ciris), white-breasted nuthatch (Sitta carolinensis), summer tanager (Piranga rubra), eastern tufted titmouse (Parus bicolor), or yellow-breasted chat (Icteria virens) between 1990 and 2014 (Table 7). However, the probability of occurrence of the grasshopper sparrow (Ammodramus savannarum) and dickcissel

(Spiza americana) decreased 9% and 6%, respectively, between 1990 and 2014 when modeled with oil price scenario 2. The slight decrease in these prairie associated species (Knopf 1994) is most likely because of the 19% increase in deciduous woodland cover associated with a decrease in oil price in 2014. The probably of occurrence of the blue-gray gnatchatcher (Polioptila caerulea), an edge associated species, also decreased about 7% between 1990 and 2014 according to all demographic-economic regression models because of the parameter coefficient deciduous woodland in the species occurrence model (Table 6). However, the probability of occurrence of the dickcissel and field sparrow (Spizella pusilla) slightly increased between 1990 and 2014 for most demographic-economic regression models. Probably of occurrence of the dickcissel was greatest when modeled with the predicted herbicide price, which resulted in the lowest woodland cover compared to models using other predicted independent variables (Table 3). Based on the species occurrence models with the greatest $\underline{R}^2$, the general shift in avian community structure towards prairie species, at the expense of woodland species, between 1966 and 1990 (Boren 1995) may continue in the rural landscape.

We observed a slightly greater difference between the probability of species occurrence modeled by logistic regression and demographic-economic regression in the urban-influenced landscape compared to the rural landscape (Table 8) because we assumed deciduous woodland and brush-treated land cover to be the same in 2014 as in 1990 (Table 3). This resulted in relatively large differences between deciduous woodland and brush-treated land cover between logistic regression models and demographic-economic regression models. In addition, there was a greater difference in the area of native grassland between logistic regression models and demographic-economic regression models in the urban-influenced landscape compared to the rural landscape (Table 3).

From 1990 to 2014 in the urban-influenced landscape there is little change in the probability of occurrence, regardless of the model used, of the chipping sparrow (Spizella passerina), common yellowthroat (Geothlypis trichas), great-horned owl (Bubo virginianus), Kentucky warbler (Oporornis formosus), northern-parula warbler (Parula americana), red-shouldered hawk (Buteo lineatus), and yellow-breasted chat (Table 8). All these species are associated with edge or forest habitat and exhibit very low probabilities of occurrence in 1990. The probability of occurrence of the American robin (Turdus migratorius) was similar between 1990 and the logistic regression model for 2014. However, the probability of occurrence was 11% lower in 2014 compared to 1990 in all demographic-economic regression models (Table 8). All demographic-economic regression models resulted in the same probability of occurrence because the area of pasture land and hay meadows, roads, and cropland, used to model the American robin, could not be projected to year 2014. We assumed each of these cover types to have the same area in 2014 as in 1990. The probability of occurrence of the common grackle (Quiscalus mexicanus), European starling (Sturnus vulgaris), and house sparrow (Passer domesticus) decrease with increasing woodland cover based on our species occurrence models (Table 6). Because the predicted rural population density resulted in a large increase in deciduous woodland cover (Table 3), the probability of occurrence of these species was lowest when modeled with the predicted variable rural population density. Based on the low probability of occurrence of forest associated birds and the slight increase for the house sparrow in 2014, previously documented shifts in avian community structure towards generalist species at the expense of woodland species, between 1966 and 1990 (Boren 1995), may continue in the urban-influenced landscape.

In addition, results from our species occurrence models are conservative for both woodland and developed species because cover of developed areas and roads could not be modeled and were held constant between 1990 and 2014. However, undoubtedly urban development will continue in urban-influenced landscape (McDonell & Pickett 1990), which will most likely result in increases of generalist and exotic species at the expense of woodland species (Galli, Leck & Forman 1976; Whitecomb et al. 1981; Terborgh 1989; Zalewski 1994).

Conservation of avian diversity is influenced greatly by the extent to which agronomic practices are applied in the landscape. Our models suggest continued intensive agriculture practices associated with urban-influenced landscapes will adversely affect native grassland bird species to a greater magnitude than extensive ranching practices in rural landscapes. Extensive management practices associated ranching enterprises appear to maintain native plant communities, which are essential for the maintenance of endemic species. Considering the tremendous increase in development and intensive agriculture practices applied at the rural-urban fringe, native vegetation will continue to be replaced with farms and introduced woodland species (Sullivan 1994). Therefore, biologists and conservationists should focus their educational programs on maintaining diversity of endemic avian species to the rural-urban fringe.

Acknowledgments

We wish to thank J.F. Stritzke, M.E. Payton, and C.E Ward for their assistance in predicting variables for regression analysis. This study was approved for publication by the Director, Oklahoma Agricultural Experiment Station and funded in part by the Department of Agronomy, Oklahoma State University.

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Table 1. Classification system used to map land use and vegetation cover types (adapted from Stoms et al. 1983).

Land use and cover type	Description
Developed area	Land occupied by residential, industrial, or other human structures and non-agricultural activities. Also includes transportation and utility facilities.
Roads	Black top, gravel, dirt roads and driveways
Water	Ponds, lakes, streams, and rivers
Cropland	Land cultivated for row crops and cereal grains but excluding grazing lands
Pasture land and hay meadows	Includes pasture land (seeded, grasslands used for grazing by cattle, sheep, goats, and horses) and hay meadows
Native grassland	Native grasslands with less than 10% cover by shrubs or trees
Deciduous forest	Vegetation dominated (>10%) by cover of broadleaf hardwoods. Mostly post oak (Quercus stellata) and blackjack oak (Q. marilandica)
Brush-treated land	Native vegetation subjected to herbicides, fire, or chaining to control woody brush encroachment
Bare ground	Land with less than 5% vegetative cover

Table 2. Summary of logistic regression models* for predicting probability of landscape cover type occurrence** in the rural and urban-influenced landscapes with year as the independent variable.

Cover type	Intercept	Parameter coefficient
Rural landscape		
Brush-treated land	4·552	-0·041(year)
Cropland	2·379	0·032(year)
Native grassland	-0·702	0·014(year)
Deciduous woodland	-0·277	0·0148(year)
Urban-influenced landscape		
Cropland	-2·943	0·068(year)
Pasture land and hay meadows	3·143	-0·026(year)
Native grassland	-0·004	0·002(year)

* Logistic regression models are for entire breeding bird survey route and separate models for each of the 50 breeding bird survey stops were used to predict probability of avian species occurrence.

** C (rank correlation between observed responses and predicted responses) was 0.50 for all logistic regression models.

Table 3. Total area (ha) of native grassland, deciduous woodland, brush-treated land, pasture land and hay meadows, and cropland in the rural and urban-influenced landscapes in year 1990 (observed), year 2014 based on logistic regression models, and year 2014 based on demographic-economic regression models.

Variable	Native grassland	Deciduous woodland	Brush-treated land	Pasture land hay meadows	Cropland
Rural landscape					
1990					
Observed	898·23	663·47	625·98	74·02	12·90
2014					
Year*	806·24	679·84	776·90	___**	9·60
Rural population density	770·07	594·03	953·71	___	___
Oil price scenario 1	791·07	673·19	853·56	___	___
Oil price scenario 2	977·59	792·77	547·44	___	___
Herbicide price	685·75	511·74	1120·32	___	___
Cattle price margin	817·94	628·28	871·84	___	___
Cattle price margin and oil price scenario 1	775·78	605·98	936·04	___	___
Cattle price margin and oil price scenario 2	908·40	618·28	790·72	___	___

Table 3. Continued.

Scenario	Native grassland	Deciduous woodland	Brush-treated land	Pasture land hay meadows	Cropland
Urban-influenced landscape					
1990					
Observed	1155·40	300·19	3·97	769·17	101·48
2014					
Year*	1069·91	___**	___	897·93	80·96
Rural population density	902·38	708·26	0·00	___	___
Number farms	1286·75	311·05	12·84	___	___
Average size farm	1245·52	365·11	0·00	___	___
Average size farm and oil price scenario 1	1304·25	302·69	3·39	___	___
Average size farm and oil price scenario 2	1188·69	421·95	0·00	___	___

* Logistic regression model with year as the independent variable.

** Total area (ha) was not calculated because of the inability of the independent variable to predict cover type change.

Table 4. Summary of demographic-economic regression models* for predicting area (ha) of landscape cover types in the rural and urban-influenced landscape with selected independent variables.

Cover type	Intercept	Parameter coefficient 1**	Parameter coefficient 2	$\underline{R}^2$
Rural landscape				
Native grassland	1919•254	-172•313(rpopd)		0•77
	1504•630	-2•150(oil)		0•82
	1396•831	-4•943(herb)		0•73
	1479•545	-0•623(cattle)		0•82
	1546•656	-0•378(cattle)	-1.294(oil)	0•99
Deciduous woodland	1648•795	-162•761(rpopd)		0•68
	1173•726	-1•378(oil)		0•83
	1155•937	-4•687(herb)		0•66
	1250•309	-0•630(cattle)		0•84
	1886•124	-0•640(cattle)	-0•581(oil)	0•93
Brush-treated land	-482•030	293•507(rpopd)		0•71
	240•746	3•533(oil)		0•71
	408•723	8•393(herb)		0•68
	250•932	1•100(cattle)		0•83
	156•599	0•756(cattle)	1•819(oil)	0•93
Urban-influenced landscape				
Native grassland	2202•160	-74•159(rpopd)		0•71
	1965•051	-0•679(nfarms)		0•63
	1221•910	0•861(asfarm)		0•73
	529•043	1•588(oil)	2•269(asfarm)	0•98
Deciduous woodland	-270•406	70•834(rpopd)		0•78
	-110•671	0•751(nfarms)		0•94

Table 4. Continued.

Cover type	Intercept	Parameter coefficient 1**	Parameter coefficient 2	$\underline{R}^2$
	763•162	-1•102(asfarm)		0•66
	1209•907	-1•024(oil)	-2•009(asfarm)	0•99
Brush-treated land	224•738	-23•132(rpopd)		0•95
	156•070	-0•220(nfarms)		0•92
	-120•242	0•381(asfarm)		0•90
	-190•772	0•162(oil)	0•525(asfarm)	0•99

* Demographic-economic regression models are for entire breeding bird survey route and were used on a per stop basis to predict probability of avian species occurrence.

** asfarm is the average farm size (ac).

cattle is the price margin for stocker cattle (U.S. $).

herb is the average price of herbicide application (U.S. $/ac).

nfarms is the number of farms.

oil is the price of crude oil (U.S. $/barrel).

rpopd is the rural population density (number people/km^2).

Table 5. Avian species responsible for shifts in avian community structure in a rural (extensively managed) and urban-influenced (extensively managed) landscape over a 24-year period, 1966 to 1990 (Boren 1995). Minor species (those that occurred three or less times) were omitted.

Species	Code	Scientific name	Type*	Habitat**	Concern***	Foraging†	Nesting††
Rural landscape							
Loss							
Yellow-breasted chat	YBCH	Icteria virens	Neotrop	Edge	High	Ground	Shrub
Blue-gray gnatchatcher	BGGN	Polioptila caerulea	Neotrop	Edge	Moderate	Canopy	Midstory
Greater roadrunner	GRRO	Geococcyx californianu	Resident	Prairie	High	Ground	Shrub
Bewick's wren	BEWR	Thryomanes bewickii	Temp	Edge	High	Ground	Cavity
Black and white warbler	BAWW	Mniotilta varia	Neotrop	Forest	Moderate	Midstory	Ground
Field sparrow	FISP	Spizella pusilla	Temp	Edge	High	Ground	Ground
Painted bunting	PABU	Passerina ciris	Neotrop	Edge	High	Ground	Shrub
Pileated woodpecker	PIWO	Dryocopus pileatus	Resident	Forest	Moderate	Bole	Cavity
Summer tanager	SUTA	Piranga rubra	Neotrop	Forest	High	Midstory	Midstory
Eastern tufted titmouse	ETTI	Parus bicolor	Resident	Forest	High	Midstory	Cavity
White-breasted nuthatch	WBNU	Sitta carolinensis	Resident	Edge	Moderate	Bole	Cavity
Gain							
Dickcissel	DICK	Spiza americana	Neotrop	Prairie	High	Ground	Ground
Wild turkey	WITU	Meleagris gallopavo	Resident	Edge	High	Ground	Ground
Barn swallow	BARS	Hirundo rustica	Neotrop	Develop	Moderate	Aerial	Other
Grasshopper sparrow	GRSP	Ammodramus savannarum	Neotrop	Prairie	High	Ground	Ground
Great-tailed grackle	GTGR	Quiscalus mexicanus	Resident	Edge	Moderate	Ground	Shrub
Little blue heron	LBHE	Egretta caerulea	Temp	Water	Moderate	Water	Shrub
Rock dove	RODO	Columba livia	Resident	Develop	Low	Ground	Other

Table 5. Continued

Species	Code	Scientific name	Type*	Habitat**	Concern***	Foraging†	Nesting††
Urban-influenced landscape							
Loss							
Black-billed cucko	BBCU	Coccyzus erythropthaim	Neotrop	Edge	High	Midstory	Shrub
Cattle egret	CAEG	Bubulcus ibis	Resident	Prairie	Low	Ground	Shrub
Yellow-breasted chat	YBCH	Icteria virens	Neotrop	Edge	High	Ground	Shrub
Chipping sparrow	CHSP	Spizella passerina	Neotrop	Forest	Moderate	Ground	Shrub
Common yellowthroat	COYE	Geothlypis trichas	Neotrop	Edge	Moderate	Ground	Shrub
Great-horned Owl	GHOW	Bubo virginianus	Resident	Edge	Moderate	Ground	Cavity
Greater prairie chicken	GPCH	Tympanuchus capido	Resident	Prairie	High	Ground	Ground
Kentucky warbler	KEWA	Oporornis formosus	Neotrop	Forest	High	Ground	Ground
Northern-parula warbler	NOPA	Parula americana	Neotrop	Forest	High	Midstory	Canopy
Red-shouldered hawk	RSHA	Buteo lineatus	Temp	Edge	Moderate	Ground	Canopy
Yellow-bellied sapsucker	YBSA	Sphyrapicus sp.	Temp	Edge	High	Bole	Cavity
Gain							
American robin	AMRO	Turdus migratorius	Temp	Develop	Low	Ground	Shrub
Gray catbird	GRCA	Dumetella carolinensis	Neotrop	Edge	High	Ground	Shrub
Common grackle	COGR	Quiscalus quiscula	Resident	Edge	Low	Ground	Midstory
Great-tailed grackle	GTGR	Quiscalus mexicanus	Resident	Edge	Moderate	Ground	Shrub
House sparrow	HOSP	Passer domesticus	Resident	Develop	Low	Ground	Cavity
Purple martin	PUMA	Progne subis	Neotrop	Develop	Moderate	Aerial	Cavity
Rock dove	RODO	Columba livia	Resident	Develop	Low	Ground	Other
European starling	EUST	Sturnus vulgaris	Resident	Develop	Low	Ground	Cavity

Table 5. Continued

Species	Code	Scientific name	Type*	Habitat**	Concern***	Foraging†	Nesting††
Wild turkey	WITU	Meleagris gallopavo	Resident	Edge	High	Ground	Ground

* Species classified as neotropical migrants (Neotrop), temperate migrants (Temp), and residents (Resident).

** Species grouped into designations of habitat occurrence: forest (Forest), forest edge and shrubland (Edge), prairie (Prairie), and developed areas (Developed).

***Species grouped into population trends: low concern (Low), moderate concern (Moderate), and high concern (High).

† Species grouped into foraging zones: open zones (Aerial), foliage 0 - 3 m (Ground), foliage 3 - 10 m (Midstory), and trunks and limbs (Bole).

†† Species grouped into nesting zones: ground (Ground), 0 - 3 m (Shrub), 3 - 10 m (Midstory), > 10 m (Canopy), cavity (Cavity), and variable heights and substrates (Other).

Table 6. Summary of logistic regression models for the rural and urban-influenced landscapes where models, with cover types as the independent variables, predict probability of occurrence of bird species on stops of the breeding bird survey route.

Species	Intercept	Parameter Coefficient 1*	Parameter Coefficient 2	Parameter Coefficient 3	Parameter Coefficient 4	Parameter Coefficient 5	C**
Rural landscape							
Barn swallow***	-0·596	0·060(wood)	-0·054(dev)^4				0·695
Bewick's wren	1·554	-0·071(wood)	-0·032(btl)	0·509(water)^2			0·750
Blue-gray gnatchatcher	3·623	-0·005(wood)^2	1·629E-6(wood)^4				0·840
Dickcissel	-1·954	0·1146(wood)					0·845
Eastern tufted titmouse	1·132	-0·113(wood)					0·834
Field sparrow	-2·402	1·043(dev)	0·287(crop)	-0·077(wood)	2·147(water)		0·840
Grasshopper sparrow	0·832	1·873(dev)	0·083(wood)				0·795
Painted bunting	1·123	-0·047(wood)					0·680
Summer tanager	2·576	-0·0016(wood)^2					0·758
White-breasted nuthatch	3·961	2·087(dev)	-0·062(wood)	0·264(dev)^3			0·804
Yellow-breasted chat	3·909	-0·066(wood)					0·746
Urban-influenced landscape							
American robin	1·706	-0·055(plhm)	-0·397(road)^2	7·50E-5(crop)^3			0·755
Chipping sparrow	6·129	-0·134(wood)	-7·142(bg)^2				0·941
Common grackle	-0·178	0·067(wood)	-0·301(road)^2				0·695
Common yellowthroat	2·483	-0·069(wood)	-0·002(crop)^2	-0·027(road)^4			0·701

Table 6. Continued.

Species	Intercept	Parameter Coefficient 1*	Parameter Coefficient 2	Parameter Coefficient 3	Parameter Coefficient 4	Parameter Coefficient 5	C**
European starling	1·988	-0·084(crop)	-1·568(road)	3·315(bg)	0·002(wood)^2	4·45E-6(crop)^4	0·788
Great-horned owl	4·153	-0·077(wood)					0·686
Greater prairie chicken	6·679	-1·528(water)					0·932
House sparrow	-0·210	-2·288(dev)	0·070(wood)	3·43E-7(grass)^4			0·732
Kentucky warbler	4·789	-0·003(wood)^2					0·773
Northern parula warbler	4·678	-0·109(wood)	-6·00E-5(crop)^3				0·868
Purple martin	3·566	-0·944(road)					0·653
Red-shouldered hawk	5·862	-0·113(wood)					0·896
Yellow-breasted chat	3·006	-1·58E-6(crop)^4					0·587

* bg is area of bare ground.

btl is area of brush-treated land.

crop is area of cropland.

dev is the area residential and industrial development.

grass is area of native grassland.

plhm is area of pasture land and hay meadows.

roads is area of roads.

water is area of water.

wood is area of deciduous forest.

** C is a rank correlation between observed responses and predicted responses.

***Refer to Table 5 for scientific name.

Table 7. Probability of occurrence of bird species modeled in year 2014 in the rural landscape modeled by logistic regression and demographic-economic regression models.

Variable	Increased 1966 to 1990			Decreased 1966 to 1990							
	BARS*	DICK	GRSP	BEWR	FISP	BGGN	PABU	WBNU	SUTA	ETTI	YBCH
1990											
Observed**	0·48	0·59	0·35	0·39	0·75	0·14	0·39	0·12	0·14	0·54	0·06
2014											
Year***	0·47	0·59	0·34	0·42	0·76	0·14	0·39	0·12	0·14	0·55	0·06
Rural population density	0·47	0·64	0·32	0·45	0·87	0·05	0·36	0·07	0·09	0·55	0·04
Oil price scenario 1	0·45	0·60	0·30	0·46	0·89	0·06	0·38	0·08	0·09	0·60	0·05
Oil price scenario 2	0·41	0·53	0·26	0·45	0·90	0·08	0·41	0·09	0·10	0·66	0·05
Herbicide price	0·50	0·69	0·35	0·45	0·86	0·04	0·35	0·06	0·08	0·51	0·04
Cattle price margin	0·46	0·63	0·31	0·45	0·88	0·05	0·37	0·07	0·09	0·57	0·04
Cattle price margin and oil price scenario 1	0·47	0·64	0·32	0·45	0·87	0·05	0·37	0·07	0·09	0·56	0·04
Cattle price margin and oil price scenario 2	0·46	0·63	0·31	0·43	0·88	0·05	0·37	0·07	0·09	0·57	0·04

* Refer to Table 5 for code and scientific name.

** Frequency of occurrence.

***Logistic regression model with year as the independent variable.

Table 8. Probability of occurrence of bird species modeled in year 2014 in the urban-influenced landscape modeled by logistic regression and demographic-economic regression models.

Variable	Increased 1966 to 1990				Decreased 1966 to 1990						
	AMRO*	COGR	EUST	HOSP	CHSP	COYE	GHOW	KEWA	NOPA	RSHA	YBCH
1990											
Observed**	0·54	0·64	0·57	0·51	0·01	0·17	0·03	0·01	0·03	0·01	0·05
2014											
Year***	0·56	0·64	0·58	0·50	0·01	0·16	0·03	0·01	0·03	0·01	0·05
Rural population density	0·43	0·43	0·41	0·48	0·01	0·19	0·04	0·01	0·04	0·01	0·05
Number farms	0·43	0·56	0·50	0·59	0·00	0·12	0·02	0·01	0·02	0·01	0·05
Average size farm	0·43	0·54	0·49	0·58	0·01	0·13	0·03	0·01	0·02	0·01	0·05
Average size farm and oil price scenario 1	0·43	0·56	0·50	0·59	0·00	0·12	0·02	0·01	0·02	0·01	0·05
Average size farm and oil price scenario 2	0·43	0·53	0·48	0·56	0·01	0·14	0·03	0·01	0·02	0·01	0·05

* Refer to Table 5 for code and scientific name.

** Frequency of occurrence.

***Logistic regression model with year as the independent variable.

2

VITA

Jon Charlton Boren

Candidate for the Degree of

Doctor of Philosophy

Thesis: EFFECTS OF AGRICULTURE INTENSIFICATION ON LANDSCAPES AND AVIAN COMMUNITY STRUCTURE

Major Field: Crop Science

Biographical:

Personal Data: Boren in Tulsa, Oklahoma, December 17, 1965, the son of Paul R. and Sue N. Boren.

Education: Graduated from Edmond Memorial High School, Edmond, Oklahoma in June 1984; received Bachelor of Science Degree in Wildlife and Fisheries Ecology from Oklahoma State University in May, 1989; received Masters of Science degree in Wildlife and Fisheries Ecology from Oklahoma State University in May, 1992; completed the requirements for Doctor of Philosophy degree in Range Ecology from Oklahoma State University in December, 1995.

Professional Experience: Philmont Boy Scout Ranch: Environmental Awareness Councilor. Bureau of Land Management: Assistant Biologist. Oklahoma State University Department of Zoology: Professional Research Technician. Oklahoma State University Department of Zoology: Teaching Assistant. Oklahoma State University Department of Agronomy: Research Assistant.

Professional Organizations: National Wildlife Federation, Oklahoma Chapter of the Wildlife Society, The Wildlife Society, Quail Unlimited, Society for Range Management, International Association for Landscape Ecology.