URBAN INFLUENCE ON DIVERSITY OF AVIFAUNA IN THE EDWARDS PLATEAU OF TEXAS: EFFECT OF PROPERTY SIZES ON RURAL LANDSCAPE STRUCTURE

A Dissertation

by

EDITH GONZALEZ AFANADOR

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

May 2006

Major Subject: Wildlife and Fisheries Sciences

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Approved by:

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ABSTRACT

Urban Influence on Diversity of Avifauna in the Edwards Plateau of Texas: Effect of Property Sizes on Rural Landscape Structure. (May 2006)

Edith González Afanador, B.S., Universidad Nacional de Colombia; M.S., Universidad Nacional de Colombia

Chair of Advisory Committee: Dr. William Grant

The urban Influence on diversity of avifauna in the Edwards Plateau ecoregion and surrounding area was studied using spatial analysis. Indices and metrics of urban influence, ownership property sizes, landscape structure, and avian diversity were calculated for 31 North American Breeding Bird Survey (BBS) transects, 12 located within the Edwards Plateau ecoregion and 18 in contiguous ecoregions. Spatial correlations were calculated between each pair of these indices.

The spatial analysis identified an emergent property at the landscape level: A "threshold of habitat fragmentation" at an ownership property size of 500 acres, which is reached when urban influence increases to an intermediate level. Highly significant spatial correlations among variables showed that property sizes lower than 500 acres produce habitat fragmentation represented by a decrease in mean patch size (MN) and proximity among habitat patches (Index PROX). Consequently, avian α -diversity (richness) decreases because both MN and Index PROX are landscape metrics related to availability of suitable habitat for avian populations.

The spatial analysis also made possible the prioritization of ecological subregions of the Edwards Plateau for conservation or restoration with respect to the threshold of habitat fragmentation and avian α and β-diversity. Balcones Canyon Lands showed a high percentage of land covered by farms smaller than 500 acres (64%), an ownership property average size above the threshold of fragmentation (1440 acres) and the highest avian α-diversity; so, management policies should focus on habitat conservation. In contrast, Lampasas Cut Plains showed the highest percentage of land covered by farms smaller than 500 acres (71%), and ownership property average size was very close to the threshold of fragmentation (625 acres); there, urban bird species are dominant and avian α -diversity is low because of the loss of native bird species. Management in this ecoregion should focus on habitat restoration. Finally, the Live Oak-Mesquite Savannah subregion showed the highest

average ownership property size (7305 acres), and the highest values of patch richness and β-diversity. Management in this ecoregion should focus on conservation of land mosaic diversity to assure native avian species turnover.

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CHAPTER I

INTRODUCTION

The Edwards Plateau ecoregion is one of the most biologically diverse areas in the State of Texas. The geological formation of the Plateau and its geographical location between the North-South transition from the North plains to the Mexican subtropics, and the East-West transition from humid plains to the southwestern deserts has resulted in a unique landscape mosaic; grassland savannas blend into the Rolling Plains to the north, a fringe of woodlands runs along the Balcones Escarpment to the East, and grasslands are interspersed with scrublands in the western and southern portions of the Plateau (Figure 1). This type of vegetation mosaic supports a biota that is both locally diverse (high α-diversity) and variable across the landscape (high β-diversity); the Edwards Plateau has been recognized as a biodiversity hotspot at both local (Hillis, 2000) and continental levels (World Wildlife Foundation, 2000; Ricketts and Imhoff 2003).

The biodiversity of the Plateau may be endangered by habitat destruction resulting from urban growth and continued fragmentation of rural properties. Some studies using birds as indicators have shown that urbanization affects landscape heterogeneity, and consequently the distribution and abundance of resources the birds use for their sustenance (Blair 2004; Donovan and Flather 2002; Luck and Wu 2002). In the first stage, at a moderate level of urban development, avian diversity reaches its peak in the area, because of the establishment of ornamental vegetation zones; the increase of between-habitat borders; and the higher water availability, all of which increase landscape heterogeneity. However, at an extreme level of urban development, avian diversity decreases as landscape heterogeneity and resource availability decrease along with the replacement of natural elements by concrete and urban structures (Whitney and Adams 1980; McKinney 2002.).

Simultaneously, urban sprawl generates economic pressure extending well past city limits into the rural landscape (Costanza et al.1997), which leads to a reduction in ownership property sizes (Adger and Luttrel 2000; Antrop 2000; Swenson and Franklin 2000; Luck and Wu 2002).

This dissertation follows the style of Landscape Ecology.

Fragmentation of large farms and ranches has been identified as the single greatest threat to wildlife habitat in Texas (Shackelford and Shackelford., 2003), and since the early 1990's the size-class distribution of rural properties has been shifting most rapidly towards smaller parcels in many areas, including the Edwards Plateau (Wilkins et al. 2003).

Reduction in property sizes leads to changes in landscape structure (Stanfield et al. 2002), i.e., changes in the spatial arrangement of land elements (Zonneveld and Forman 1989; Baudry 1993), which commonly leads to changes in biodiversity (Donovan and Flather 2002; Lovett-Doust et al. 2003). This has been especially prominent along the Austin-San Marcos-San Antonio metropolitan belt, that is, in the eastern border of the Edwards Plateau.

Among the more pervasive landscape changes whose impact on biodiversity is potentially great, yet remains poorly understood, if is that associated with changes in ownership property sizes along the urban – rural gradient. This research focuses on identifying urban influence on diversity of avifauna in the Edwards Plateau and surrounding area, by examining the spatial relationship among urban influence, ownership property sizes, landscape structure and avian diversity. I have used the conceptual framework of landscape ecology (Forman 1984; Zonneveld and Forman 1989; Turner 1989; McGarigal and Marks 1995; Turner et al. 2001; Gergel and Turner 2002) and the quantitative tools of spatial statistics (Isaaks and Srivastava 1989; Legendre and Fortin 1989; Fortin and Gurevitch 1993; Dutilleul et al.1993; Dale and Fortin 2002; Fortin and Payette 2002) with which to do so.

CHAPTER II

IDENTIFICATION OF CONSERVATION PRIORITY ZONES IN THE EDWARDS PLATEAU ECOREGION USING ALPHA AND BETA DIVERSITY OF AVIFAUNA AS INDICATORS

Introduction

The Edwards Plateau is one of the most biologically diverse areas in the State of Texas. The geological formation of the Plateau and its geographical location between the North-South transition from the North plains to the Mexican subtropics, and the East-West transition from humid plains to the southwestern deserts has resulted in a unique landscape mosaic; grassland savannas blend into the Rolling Plains to the north, a fringe of woodlands runs along the Balcones Escarpment to the East, and grasslands are interspersed with scrublands in the western and southern portions of the Plateau (Figure II.1). This type of vegetation mosaic supports a biota that is both locally diverse (high α-diversity) and variable across the landscape (high β-diversity); the Edwards Plateau has been recognized as a biodiversity hotspot at both local (Hillis, 2000) and continental levels (World Wildlife Foundation 2000; Ricketts and Imhof 2003).

The biodiversity of the Plateau may be endangered by habitat degradation resulting from urban growth and continued fragmentation of rural properties. Fragmentation of large farms and ranches has been identified as the single greatest threat to wildlife habitat in Texas (Shackelford and Shackelford 2003), and since the early 1990's the size-class distribution of rural properties has been shifting most rapidly towards smaller parcels in many areas, including the Edwards Plateau (Wilkins et al. 2003). Reduction in property sizes leads to changes in landscape structure (Stanfield et al. 2002), i.e., changes in the spatial arrangement of land elements (Zonneveld and Forman 1989; Baudry 1993), which commonly leads to changes in biodiversity (Donovan and Flather 2002; Lovett-Doust et al. 2003).

Although the priorities for biodiversity conservation within the Edwards Plateau ecoregion have been declared at continental (Ricketts and Imhof 2003), local (Hillis 2000), and regional levels, few studies can be found in the scientific literature. Processes that produce habitat destruction, changes in land use, changes in ownership property size, and urbanization occur at all of these levels. Therefore, it is very important to identify natural regions with different values of biodiversity in order to define conservation priorities at a regional level. In this chapter, the natural avian subregions were identified using avian abundance and species composition. Second, the conservation priority areas were identified using alpha and beta avian diversity.

Study area

The study area was bounded by a circle with radius 300 km centered at the geographical center point of the Edwards Plateau Ecoregion, including all ecological subregions of the Edwards Plateau and portions of ecological subregions for 5 ecoregions around the Edwards Plateau: South Texas Brush, Blackland Prairie, Llano Uplift, Rolling Plains, and Oak Woods (Figure II.1).

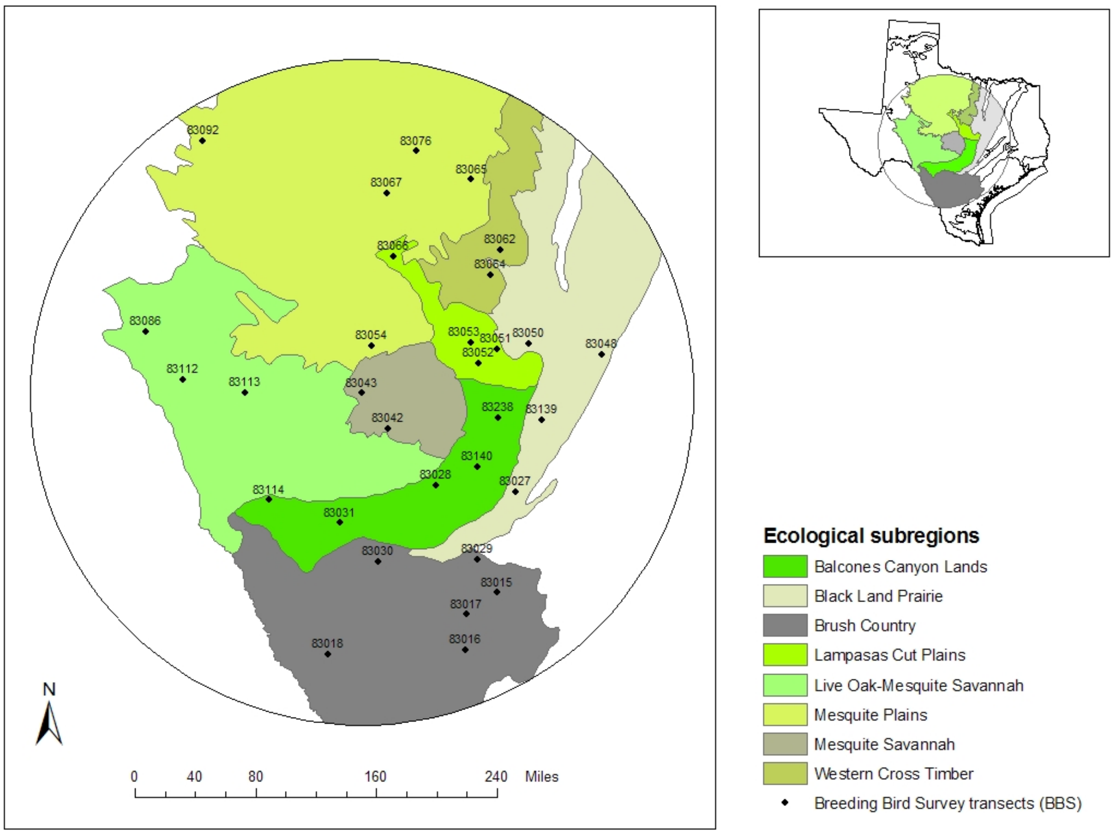

Figure II.1. *Geographic location of the study area. The study area included the 8 ecological sub-regions referred to in this dissertation. Locations of the 31 North American Breeding Bird Survey (BBS) transects included in the present study also are indicated; numbers correspond to the BBS code, of which the first two digits (83) signify Texas, and the final three digits (015-238) correspond to the transect number.*

The climate ranges from subtropical steppe to subtropical sub-humid, with mean annual precipitation ranging from 375 mm in the west to 750 mm in the east, about three-fourths of which falls during the growing season (April through mid-November).

The area is predominantly shrub land grazed by cattle, sheep, and goats; but local tracts are cultivated for domestic pasture and hay, cotton and grain sorghum (grown locally on irrigated land), and some pecan orchards in flood plains. Landowners commonly lease their land for hunting deer, quail, mourning dove, wild turkey, and javelina. Many rural areas are experiencing greatly increased residential development, especially in the eastern portion of the region, due in large part to the influence of large cities such as San Antonio and Austin (Wilkins et al. 2003).

Methods

First, I describe the databases, followed by the description of the calculations for the identification of the natural avian regions, then the description of the calculations for avian diversity indices, and lastly the ecological characterization of communities within the natural avian regions and the conservation priorities identified for each of the natural avian regions.

Description of database

I collected avian data from 1990 to 1994 from 31 North American Breeding Bird Survey (BBS) transects (ranging in length from 35.7 to 39.3 km), including 12 transects located within the three ecological subregions of the Edwards Plateau Ecoregion (Live Oak-Mesquite Savannah, Balcones Canyon Lands, and Lampasas Cut Plains) and the 19 closest transects located in adjacent ecological subregions: four in Blackland Prairie, two in Mesquite Savannah, two in Western Cross Timber, five in the Mesquite Plains, and six in the Brush Country (Figure 1). Data were not available for the ecological subregions of the Trans Pecos Ecoregion for all years in some of the 31 transects, but each transect had at least 3 consecutive years of data. Transects in ecological regions adjacent to the Edwards Plateau ecoregion were chosen to represent an "ecological border" (as defined by Cadenasso et al 2003). Data from 1990 to 1994 were included for two reasons: first, land cover data for the Edwards Plateau were available only for 1992; and, second, the landscape of the Edwards Plateau was changing rapidly during this period (Wilkins et al. 2003; see Introduction).

The Breeding Bird Survey (BBS), which began in 1966, consists of a set of roadside surveys (over 3,500 transects have been established) conducted each June by experienced birders to provide an index of population change for songbirds (Sauer et al. 2003). Data include the number of species of birds and the number of individuals of each species observed on each transect. Each species is characterized in terms of habitat preferences (e.g., grassland) and migration status (e.g., neotropical migrant).

Identification of natural avian regions

Natural avian regions were identified based on similarity of avian abundance and species composition among the BBS transects using the clustering analysis method described by Ward (1963). The analysis included only those species that appeared in at least 10 of the 31 transects. The clustering analysis procedure consisted of first constructing a 31x31 resemblance matrix containing the Euclidian distances (D) between each pair of transects:

$$D_{jk} = \sqrt{\textstyle\sum_{i=1}^{S} (X_{ij} - X_{ik})^2} \qquad \text{(II.1)}$$

where D_{jk} represents the Euclidian distance between transects j and k, X_{ij} and X_{ik} represent the logarithm of the number of individuals of the i^{th} species observed per 10 km of transect on transects j and k, respectively, while S represents the number of species included in the analysis. The two transects with the lowest D were then clustered together, and a new D was calculated between this cluster and each of the other transects:

$$D_{(jk),h} = \alpha 1 * D_{jh} + \alpha 2 * D_{kh} + \beta * D_{jk} \qquad \text{(II.2)}$$

where $D_{(jk),h}$ represents the Euclidian distance between the newly formed cluster (transects j and k) and transect h, and α1, α2, and β are weighting factors representing the number of transects (n) involved in calculating the D values for transects j, k, and h, respectively ($\alpha 1 = (n_j + n_h) / (n_j + n_k + n_h)$, $\alpha 2 = (n_k + n_h) / (n_j + n_k + n_h)$, and $\beta = n_h / (n_j + n_k + n_h)$). The program ADE-4 (Thioulouse et al., 1997) was used to conduct the clustering analysis and to generate a dendrogram that summarizes the results. Each of the natural avian regions was characterized based on the BBS designations of habitat preference and migration status of the species belonging to it.

Avian diversity indexes

The abundance (total number of individuals of all species per 10 km of transect), two indexes of α-diversity (species richness (number of different species per 10 km of transect) and Shannon's diversity (H')), and an index of β-diversity (1 - S) (Ludwig and Reynolds 1988), each averaged over the period from 1990 to 1994, were calculated for each transect. Shannon's diversity (H') was calculated using the following equation:

$$H' = -\sum \left(\frac{n_i}{N} \right) * \ln \left(\frac{n_i}{N} \right) \qquad \text{(II.3)}$$

where n_i represents the number of individuals of the ith species per 10 km of the transect and N represents the total number of individuals of all species per 10 km of the transect. The index of β-diversity was calculated using the following equation:

$$\beta\text{-diversity} = 1 - S \qquad \text{(II.4)}$$

$$\textit{where } S = (2 * pN) / (aN + bN) \qquad \text{(II.5)}$$

and aN and bN represent the total number of individuals of all species per 10 km of transect in the first and second transects, respectively, and pN represents the sum of the lower of the 2 abundances for each of the species that occur on both transects; for example, if species X, Y, and Z are represented by 2, 4, and 8 individuals, respectively, on the first transect and 3, 5, and 7 individuals, respectively, on the second transect, pN would equal 13 (2 + 4 + 7). The mean and standard deviation of abundance, species richness, H', and β-diversity of transects in each of the natural avian regions were also calculated. Finally, bird species were grouped by breeding habitat and by migration status, as identified in the BBS, and mean abundance and richness were calculated for each of the groups for each of the natural avian regions.

Ecological characterization of communities within natural avian regions.

Each of the natural avian regions was characterized using the results of the average of avian abundance, species richness, H', and β-diversity. Then the conservation priorities were identified according to the diversity result for each natural avian region.

Results

Natural avian regions

Results of cluster analysis identified five natural avian regions (A through E), which emerged at dissimilarity levels ≤ 0.45 (Figure II.2).

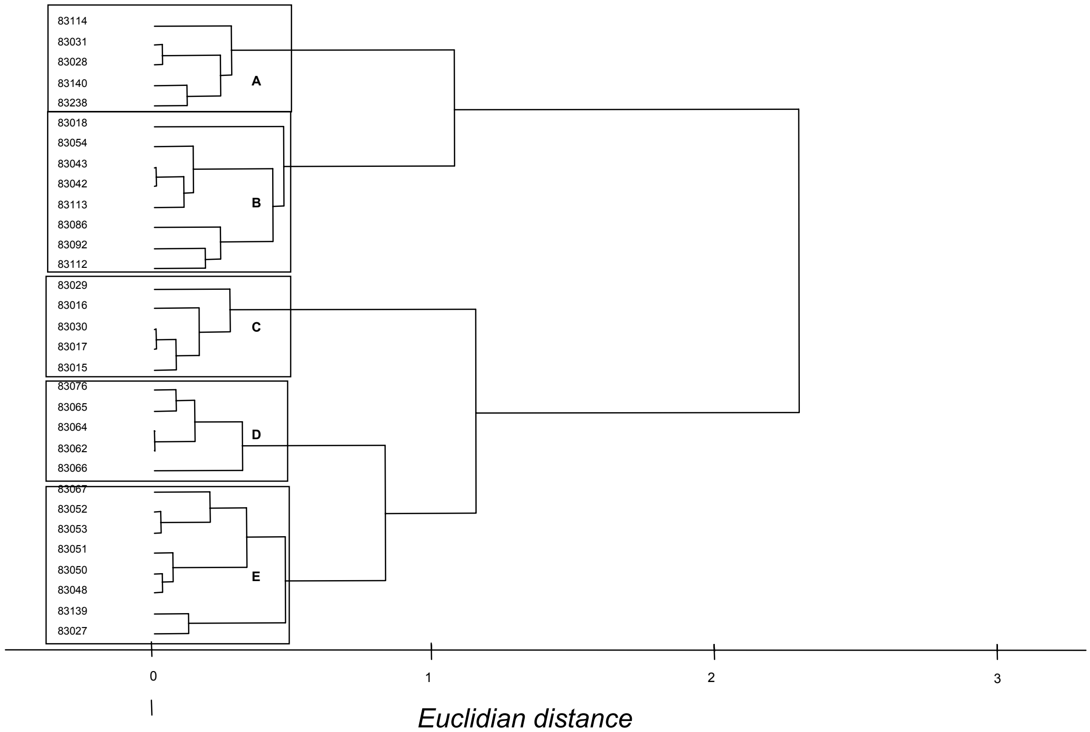

Figure II.2. *Results of cluster analysis identifying five natural avian regions (A through E). Regions based on dissimilarity of avian abundance (total number of individuals of all species per 10 km of transect) and species composition of the 31 North American Breeding Bird Survey transects from which data were drawn for the present study.*

Region A consists of five transects in Balcones Canyon Lands; region B includes 3 transects in the Live Oak Mesquite Savannah, 3 in the Mesquite Savannah, 1 in the Brush Country, and 1 in the Mesquite Plains; region C consists of 5 transects in the Brush Country; region D includes 2 transects in the Mesquite Plains, 2 in the Western Cross Timbers and 1 transect in Lampasas Cut Plains; finally, region E includes 4 transects in Lampasas Cut Plains and three transects in the Black Land Prairie ecoregion (Figure. 3). Interestingly, the natural avian regions that are the most dissimilar (A and E) are the closest to each other geographically. Natural avian regions seem to correspond to the ecological sub-regions of the study area (Figure. II.3).

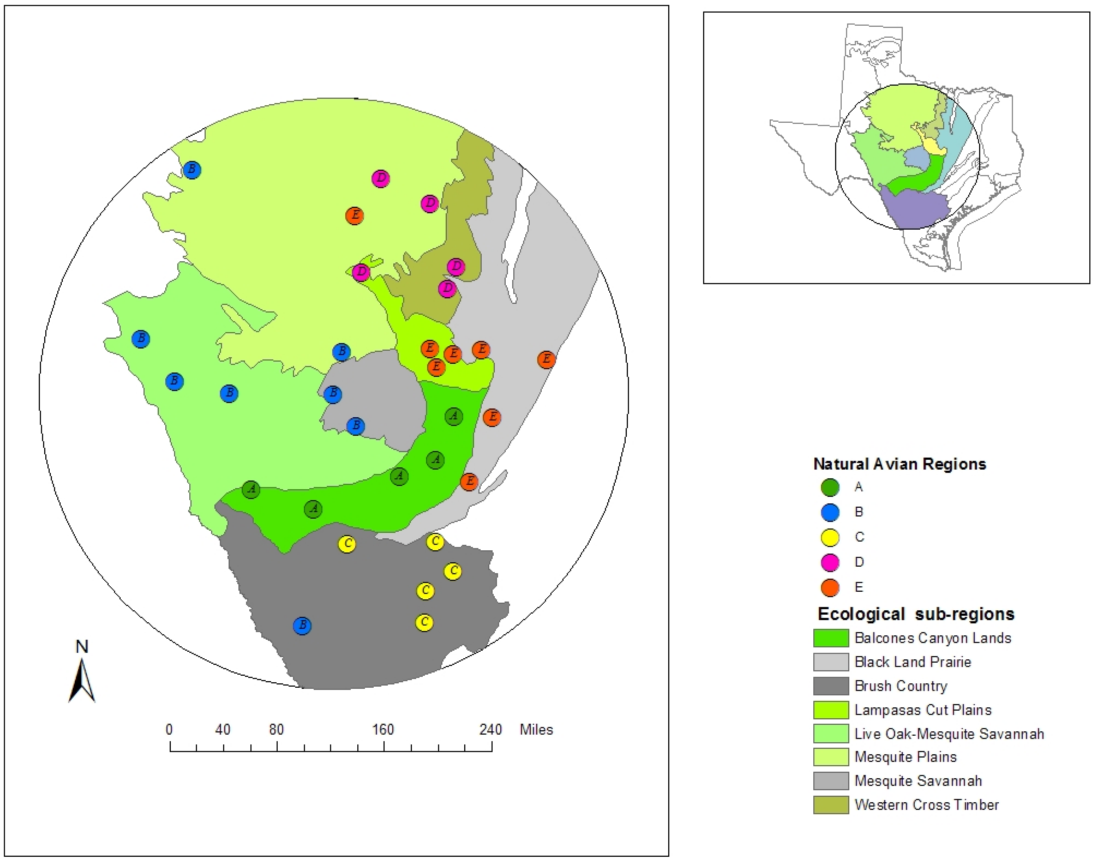

Figure II.3. *Natural avian regions and ecological sub-regions in the study area.*

Avian diversity indexes

Avian abundance on the 31 transects ranged from 68 to 385 individuals (Table A-1), with higher abundances tending to occur on the northern and southern-most transects (transect numbers 76, 66, 15, and 29, Figure II.1). Mean abundance was highest for natural avian region D, followed by regions C, E, A, and B, respectively (Table II.1).

Species richness on the 31 transects ranged from 36 to 71 species (Table A-1), with higher species richness tending to occur along an East-West band of transects lying midway between the northern and southern extremes of the study area (transect numbers 140, 28, 114, and 31, Figure II.1). Mean species richness was highest for natural avian region A, followed by C, with B, D, and E having the same, relatively lower, richness (Table II.1).

Species diversity, based on values of H’, on the 31 transects ranged from 1.1 to 3.6 (Table A-1), with no obvious geographical gradients within the study area. Mean species diversity was highest for natural avian region A, followed by E, B, D, and C, respectively (Table II.1).

Table II.1. *Mean (±1SD) avian abundance (total number of individuals of all species per 10 km of transect), two indices of α diversity, species richness (number of different species per 10 km of transect), species diversity (H'), and an index of β-diversity (1-S). β-diversity is based on Sorensen's similarity index (S). Indices are calculated for the five natural avian regions (Figs. 1 and 2) identified in the present study*

Natural avian regions	A	B	C	D	E
Avian abundance	178 (27)	166 (50)	269 (43)	275 (111)	184 (69)
Species richness	15 (2.0)	11 (1.7)	13 (2.0)	11 (1.7)	11 (2.0)
Species diversity H'	3.4 (0.21)	2.6 (0.89)	1.4 (0.17)	2.5 (0.73)	2.8 (0.70)
β -diversity (1-Sorensen)	1.6 (0.3)	2.2 (0.3)	1.5 (0.2)	1.5 (0.2)	2.0 (0.3)

Beta-diversity based on mean values of S was highest for natural avian region B, followed by E, A, C, and D, respectively; based on mean values of S', β -diversities were lower (S values higher), and regions A and E switched positions (Table II.1).

Ecological characterization of communities within natural avian regions

An ecological characterization of the five natural avian regions, based on the BBS designations of habitat preference and migration status of the species belonging to each, is presented in Table II.2, as well as a list of the species observed on each transect, arranged by habitat preference, in Table A-2.

Table II.2. *Ecological characterization of the five natural avian regions identified in this study. Characterization is based on the North American Breeding Bird Survey designations of habitat preference and migration status of the species belonging to each. Table entries represent the number (percent) of species belonging to the indicated category.*

			Habitat preference			Migration status	
Natural avian region	Woodland	Scrubland	Grassland	Urban	Neotropical migrant	Short distance migrant	Permanent resident
A	17(27)	19 (27)	3(16)	12(26)	23 (32)	9 (21)	19 (23)
B	9 (15)	19 (27)	5 (26)	9 (20)	13 (18)	12 (29)	17 (20)
C	14 (23)	15 (21)	4 (21)	9 (20)	11 (15)	9 (21)	22 (26)
D	10 (16)	8 (11)	4 (21)	7 (15)	11 (15)	6 (14)	12 (14)
E	12 (19)	9 (13)	3 (16)	9 (20)	13 (18)	6 (14)	14 (17)
Total	62	70	19	46	71	42	84

Discussion

In the study area, there are five conservation priority zones corresponding to the 5 natural avian regions, identified using cluster analysis. These areas are very coincidental with the ecological sub-regions (Wu et al. 2002). The characteristics in avian diversity are different within each natural avian region, which creates a conservation priority order as the natural avian region A first followed by B, E, C, and D.

Natural avian region A includes all transects located in the Balcones Canyon Lands subregion, which is located in the southeast portion of the Edwards Plateau ecoregion (Figure II.3). This natural avian region presented the highest values of species richness, composed by high number of neo-tropical migrants (23) of woodland and scrubland habitats. It is important to prioritize monitoring the effects of urban growth on avian diversity because in this natural avian region all urban species that were registered were found in the study area (12), which can indicate that those species have replaced native species of woodland and scrubland habitats (Table II.2).

Natural avian region B includes transects located in Live Oak-Mesquite Savannah in Edwards Plateau ecoregion and Mesquite Savannah in Llano Uplift ecoregion (Figure II.3). This natural avian region had the highest value of beta-diversity indicating that all transects included in this region, had a high number of species, but different ones, which indicates a high turnover of species. Conservation in this region has to be focused on habitat for short distance migratory birds of scrubland and grassland (Table II.2).

Natural avian region E has the lowest species richness in the Edwards Plateau ecoregion (11). Urban species make an important contribution to the alpha-diversity and high value of beta-diversity, showing a high replacement of native species by urban species. Natural avian region C and D supported the highest values of avian abundance, but lowest values of Shannon index and beta-diversity. In region C conservation of scrubland habitat, especially for native permanent resident species is a priority.

Finally, it is important to say that the east portion of the study area has a strong urban influence because of the Austin-San Marcos-San Antonio metropolitan belt. It has introduced new environmental conditions, which have enriched the avian diversity of species adapted to urban conditions. However, if native habitat transformation increases, urban species will likely become dominant and replace native avian species in the area. Contiguous natural avian regions A with higher values of richness and E with lowest value, may be showing that A in 1992 was in an intermediate grade of urbanization, but E had reached a threshold of urbanization in which environmental conditions became not suitable for native species.

These results seem to be in agreement with Blair (2004) who found that, in Ohio, maximum values of diversity at intermediate levels of urbanization decreased when urbanization level increased. Because environmental conditions change, urbanization facilitates the replacement of native species by species adapted to urban conditions, e.g. Mooring Dove, which is not good for an area that has been declared a hot spot of biodiversity (Ricketts and Imhof 2003; WWF 2000; Hillis 2000).

Conclusions

In the study area there are five natural avian regions based on cluster analysis. Those natural avian regions are very similar with ecological subregions recognized by Wu et al (2002).
Urban influence along the eastern border of the Edwards Plateau ecoregion introduced changes in environmental conditions in 1992, which had enriched avian diversity in natural avian region A (Balcones Canyon Lands subregion) due to the introduction of 13 species of urban birds. However, in contiguous region E (Lampasas Cut Plains and Black Land Prairies subregions), the urbanization level had reached a threshold in which, urban bird species became dominant and bird diversity was lowered due to the loss of native bird species.

It is important to monitor increases in urbanization and habitat transformation in natural avian region A, because it is the natural avian region with the highest avian diversity, and is especially rich in woodland and scrubland species. Subregion A is important to conservation of habitats for neotropical migrants.

Conservation plans for natural avian region B (Live Oak-Mesquite Savannah and Mesquite Savannah) have to be focused on the preservation of habitat diversity for the interchange of native bird species. High values of β-diversity and species composition showed a high turnover between native species.

CHAPTER III

SPATIAL RELATIONSHIPS BETWEEN LANDSCAPE STRUCTURE AND AVIAN DIVERSITY IN THE EDWARDS PLATEAU OF TEXAS: IMPLICATIONS FOR CONSERVATION PLANNING

Introduction

The Edwards Plateau of Texas is one of the areas of highest biological diversity in the State. The geological formation of the Plateau and its geographical location between the north-south transition from north plains to the Mexican subtropics, and the east-west transition from humid plains to the southwestern deserts has resulted in a unique landscape mosaic; grassland savannas blend into the Rolling Plains to the north, a fringe of woodlands runs along the Balcones Escarpment to the east, and grasslands are interspersed with scrublands in the western and southern portions of the Plateau (Figure. 1). This vegetation mosaic supports a biota that is both locally diverse (high α-diversity) and variable across the landscape (high β-diversity); the Edwards Plateau has been recognized as a biodiversity hotspot at both local (Hillis 2000) and international levels (World Wildlife Foundation 2000; Ricketts and Imhof 2003).

However, the biodiversity of the Plateau may be endangered by shifts in the landscape mosaic resulting from continued fragmentation of rural properties. Fragmentation of large farms and ranches has been identified as the single greatest threat to wildlife habitat in Texas (Shackelford and Shackelford 2003), and since the early 1990's the Edwards Plateau has been among the areas in which the size-class distribution of rural properties has been shifting most rapidly towards smaller parcels (Wilkins et al. 2003). Reduction in property sizes leads to changes in landscape structure (Stanfield et al. 2002), that is, changes in the spatial arrangement of land elements (Zonneveld and Forman 1989; Baudry 1993), which commonly leads to changes in biodiversity (Donovan and Flather 2002; Lovett-Doust et al. 2003).

As the fragmentation of rural properties continues, conservation planning must be based on a more thorough understanding of the linkages between specific elements of landscape structure and specific aspects of biological diversity. We now have both the conceptual framework of landscape ecology (Forman 1984; Zonneveld and Forman 1989; Turner 1989; McGarigal and Marks, 1995; Turner et al. 2001; Gergel and Turner 2002) and the quantitative tools of spatial statistics (Isaaks and Srivastava 1989; Legendre and Fortin 1989; Fortin and Gurevitch 1993; Dutilleul et al.1993; Dale and Fortin 2002; Fortin and Payette 2002) with which to do so. In this chapter, I investigate specific linkages between landscape structure and avian diversity within the Edwards Plateau of Texas. I first test for statistically significant spatial correlations among 6 indexes of landscape structure and 4 indexes of avian diversity, and then explore the implications of these results for conservation planning.

Study area

The study area was bounded by a circle of radius 300 km centered at the geographical center point of the Edwards Plateau Ecoregion, which includes all of the Edwards Plateau and portions of 5 other ecoregions--South Texas Brush, Blackland Prairie, Llano Uplift, Rolling Plains, and Oak Woods.

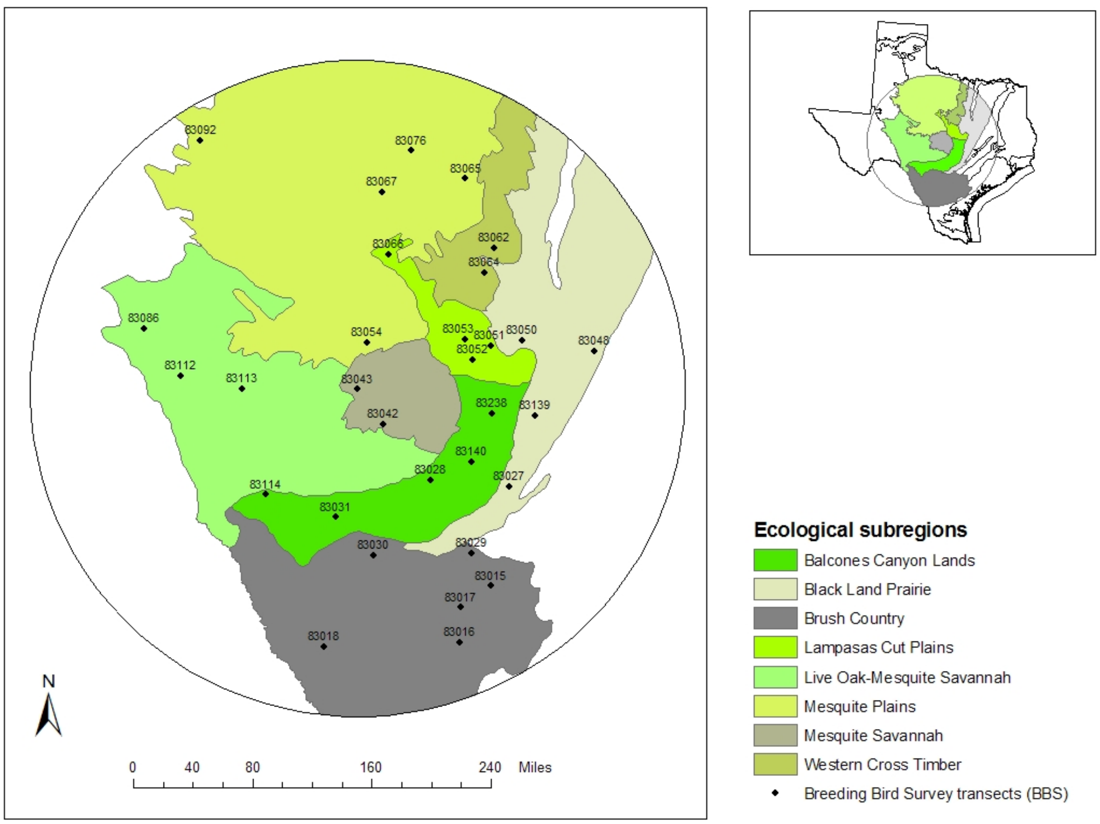

Figure III.1. *Geographic location of the study area. The study area included the 8 ecological sub-regions referred to in this dissertation. Locations of the 31 North American Breeding Bird Survey (BBS) transects included in the present study also are indicated; numbers correspond to the BBS code, of which the first two digits (83) signify Texas, and the final three digits (015-238) correspond to the transect number.*

For purposes of the present study, I used the 8 Texas ecological subregions (Gould 1975. adapted by Wu et al. 2002), included in these ecoregions: Balcones Canyon Lands, Black Land

Prairies, Brush Country, Lampasas Cut Plains, Live Oak-Mesquite Savannah, Mesquite Savannah, Mesquite Plains, and Western Cross Timbers (Figure III.1).

The climate ranges from subtropical steppe to subtropical sub-humid, with mean annual precipitation ranging from 375 mm in the west to 750 mm in the east, about three-fourths of which falls during the growing season (April through mid-November). The area is predominantly shrub land grazed by cattle, sheep, and goats, but local tracts are cultivated for domestic pasture and hay; cotton and grain sorghum are grown locally on irrigated land and there are some pecan orchards on flood plains. Landowners commonly lease their land for hunting deer, quail, mourning dove, wild turkey, and/or javelina. Many rural areas are experiencing greatly increased residential development, especially in the eastern portion of the region, due in large part to the influence of large cities such as San Antonio and Austin (Wilkins et al. 2003).

Methods

I first describe the databases and the calculation of the indexes used to represent avian diversity and landscape structure. Second, I describe the test for spatial autocorrelation for each variable (index). I then describe the tests for spatial correlations between each pair of variables. Finally, I describe the test for correlation between the values of each pair of variables, corrected for effects of spatial correlation.

Avian diversity indexes

I first assembled avian data collected from 1990 to 1994 from 31 North American Breeding Bird Survey (BBS) transects (ranging in length from 35.7 to 39.3 km), including 12 transects located within the three ecological subregions of the Edwards Plateau Ecoregion (Live Oak-Mesquite Savannah, Balcones Canyon Lands, and Lampasas Cut Plains) and the 19 closest transects located in adjacent ecological subregions: 4 in Blackland Prairie, 2 in Mesquite Savannah, 2 in Western Cross Timber, 5 in the Mesquite Plains, and 6 in the Brush Country (Fig. III.1); no data were available for ecological subregions of the Trans Pecos Ecoregion, and data were not available for all years on some 31 transects, but each transect had at least 3 consecutive years of data. I chose transects in adjacent ecoregions to represent an “ecological border” (as defined by Cadenasso et al. 2003). I only included data from 1990 to 1994 for two reasons; land cover data for the Edwards Plateau only were available for 1992 (see Section 3.2), and the landscape of the Edwards Plateau was changing rapidly during this period (Wilkins et al. 2003).

The BBS, which began in 1966, consists of a set of roadside surveys (over 3,500 transects have been established) conducted each June by experienced birders to provide an index of population change for songbirds (Sauer, Hines, and Fallon, 2003). Data include the number of species of birds and the number of individuals of each species observed on each transect. Each species also is characterized in terms of habitat preferences (e.g., grassland) and migration status (e.g., neotropical migrant).

I then calculated, for each transect, abundance (total number of individuals of all species per 10 km of transect), two indexes of α-diversity (species richness (number of different species per 10 km of transect) and Shannon's diversity (H')), and an index of β-diversity (1 - S) (Ludwig and Reynolds., 1988), each averaged over the period from 1990 to 1994.

$$H' = -\sum\left(\frac{n_i}{N}\right) * \ln\left(\frac{n_i}{N}\right) \qquad \text{(III.1)}$$

where n_i represents the number of individuals of the ith species per 10 km of the transect and N represents the total number of individuals of all species per 10 km of the transect.

$$\beta\text{-diversity} = 1 - S \qquad \text{(III.2)}$$

$$\text{where } S = (2 * pN) / (aN + bN) \qquad \text{(III.3)}$$

and aN and bN represent the total number of individuals of all species per 10 km of transect in the first and second transects, respectively, and pN represents the sum of the lower of the 2 abundances for each of the species that occur on both transects; for example, if species X, Y, and Z are represented by 2, 4, and 8 individuals, respectively, on the first transect and 3, 5, and 7 individuals, respectively, on the second transect, pN would equal 13 (2 + 4 + 7). I also calculated the mean and standard deviation of abundance, species richness, H', and β-diversity of the transects in each of the 5 ecological sub-regions. Finally, I grouped bird species by breeding habitat and by migration status, as identified in the BBS, and calculated mean abundance and richness for each of the groups for each of the 8 ecological sub-regions.

Landscape structure indexes

I obtained land cover data for the Edwards Plateau Ecoregion for 1992 from the National Land Cover Dataset (NLCD) of the United States Geological Service (USGS); more recent data were not available. The spatial resolution of the data is 30 meters, mapped in the Albers Conic Equal

Area projection, NAD 83. I first regrouped the 21-class land cover classification scheme of the NLCD into 10 land cover classes, using the Spatial Analyst tool of Arc 8 GIS (ESRI 2000). I then identified which of these more aggregated land cover classes provide habitat for the avian species seen on the BBS transects included in this study.

I then created, using the Buffer Wizard tool in Arc 8 GIS (ESRI 2000), buffer scenes of 5, 10, and 20 km around the 31 BBS transects (buffer shape files). I used each buffer shape file to "cut" an identical area on the NLCD using the Spatial Analyst tool; I used these scenes to relate landscape structure to avian diversity (Donovan and Flather 2002).

I then calculated, within each of the 31 buffer scenes, 6 indexes of landscape structure, 2 at the landscape mosaic level, and 4 at the land cover class level. At the landscape mosaic level, I calculated richness of patches (PR), which is the number of types of land cover classes present in the landscape of each buffer scene, excluding the landscape border if present, and Shannon Diversity (SHDI):

$$SHDI = -\sum_{i=1}^{m} P_i * \ln P_i \qquad \text{(III.4)}$$

where P_i is the proportion of the landscape of each buffer scene occupied by land cover class i.

At the land cover class level, I calculated for each of the land cover classes, percent of land PL, patch density PD (number of patches / 100 ha), mean patch size MN (ha), and the proximity index (PROX), using FRAGSTATS 2.0 (McGarigal and Marks 1995; Stanfield et al. 2002).

$$PROX = \sum_{s=1}^{n} \frac{a_{ijs}}{h_{ijs}^2} \qquad \text{(III.5)}$$

where PROX represents the proximity index for focal patch i, a_{ijs} is the area (m^2) of patch ijs within a specified neighborhood (m) of patch ij, and h_{ijs} is the distance (m) between patch ijs and patch ijs, based on patch edge-to-edge distance, computed from cell center to cell center. I selected a 100 meter search radius under the assumption that this was within the daily range of movements of all of the avian species included in this study.

Low index values indicate patches that are relatively isolated from other patches within the specified buffer distance, and high values indicate patches that are relatively connected to other patches (Turner et al, 2001). I used PROX as an indicator of wildlife habitat fragmentation; high values indicate less fragmentation and low values indicate more fragmentation (Mortberg

2001; Brooks et al. 2002). I also calculated the mean and standard deviation of PR, SHDI, PL, PD, MN, and PROX of the buffer scenes in each of the 5 ecological sub-regions.

Spatial autocorrelation of variables

I tested for spatial autocorrelation in each of the indexes of avian diversity and landscape structure using Mantel Tests (Fortin et al., 1993). The Mantel Test (r) is a regression in which the variables themselves are distance, or dissimilarity (ecological distances), matrixes summarizing pair-wise similarities among sample locations:

$$r = \frac{\sum\sum stdA_{ij} * stdB_{ij}}{n-1} \text{ (sum from i to n and sum from j to n, for } i \neq j) \qquad \text{(III.6)}$$

where n is the number of sample locations, i and j identify the matrix element, B_{ij} is the Euclidian distance matrix of location points, and A_{ij} is the dissimilarity matrix of the variable of interest, in the present case, avian abundance, species richness, H', PR, SHDI, PL, PD, MN, or PROX. For Mantel test calculations, I used the PASSAGE program (Rosenberg 2005).

Spatial correlations between pairs of avian diversity and landscape structure indexes

To determine if the indexes of avian diversity and landscape structure are spatially correlated, I conducted a Cross Mantel Test (r) (Fortin and Gurevitch 1993), between each pair of avian diversity and landscape structure indexes. This set of analyses tests for a spatial relationship *per se* between differences of pairs of values of both variables (avian diversity and landscape structure), but does not indicate the degree of correlation between the values of the two variables. For example, if the differences in the values of variable A and the differences in the values of variable C increase or decrease with increasing distance (that is, with increasing geographical distance between the points where the values of the variables were measured), the variables A and C are positively spatially correlated. If the differences in the values of variable A increase with distance and the values of variable C decrease with distance, or vice versa, then variables A and C are negatively spatially correlated.

$$r = \frac{\sum\sum stdA_{ij} * stdC_{ij}}{n-1}$$ (sum from i to n and sum from j to n, for i ≠ j) (III.7)

where n is the number of sample locations, i and j identify the matrix element, A_{ij} is the dissimilarity matrix of one of the variables of interest, that is, an index of avian diversity (species richness, avian abundance, or species diversity), and C_{ij} is the dissimilarity matrix of the other variable of interest, that is, an index of landscape structure (PR, SHDI, PL, PD, MN, PROX).

Correlation between values of pairs of avian diversity and landscape structure indexes

I conducted a Pearson's pair-wise correlation (ρ) between each of the avian diversity and landscape structure indexes. This set of analyses identifies the degree of correlation between the values of the two variables,

$$\rho = \frac{\sum_{i=1}(u_i - m_u)(v_i - m_v)}{S_u S_v} \qquad \text{(III.8)}$$

where u and v are two variables (u is one of the 4 avian diversity indexes and v is one of the 5 landscape structure indexes). m_u and m_v are their respective means, and S_u and S_v are their respective standard deviations. I then used the Modified *t*-test for autocorrelation (CHR), which corrects the degrees of freedom based on the amount of autocorrelation in the data, to assess the correlation between each pair of spatially correlated variables (Clifford et al. 1989; Dutilleul 1993). The procedure calculates the amount of spatial autocorrelation of variables to determine how different the effective sample size (n') is from the number of observations.

$$n'(R) = \frac{n^2}{\sum\sum cor(u_i u_j)} \qquad \text{(III.9)}$$

where R is the autocorrelation matrix, n is the number of observations, and u_i, and u_j are the observations of the two variables. The corrected degrees of freedom (n'-2) were then used to test the significance of the correlation; I used the PASSAGE program (Rosenberg 2005) to perform these calculations.

Results

Avian diversity indexes

Ninety-two different species, including species classified by breeding habitat as woodland, scrubland, grassland, wetland, and urban breeders, and by migration status as neotropical migrant, short-distance migrant, and permanent resident, were reported from 1990 to 1994 on the 31 BBS transects (Table A-3).

Species representing each of these breeding habitats and migration statuses were reported on virtually all transects (Table A-2). Scrubland- and urban-breeding birds were relatively more abundant and woodland, grassland, and wetland breeding birds were relatively less abundant (Table III.1A), and migratory birds were relatively more abundant and permanent residents were relatively less abundant (Table III.1B), in all 8 ecological sub-regions. The number of species representing these different breeding habitats and migratory statuses generally followed the same patterns as abundance, although the number of woodland-breeding species is essentially as high as the number of scrubland- and urban-breeding species (Table III.1).

Avian abundance on the 31 transects ranged from 68 to 385 individuals (Table A-1), with higher abundances tending to occur on the northern- and southern-most transects (transect numbers 76, 66, 15, and 29, Figure 4). Mean abundance was highest in Mesquite Plains, followed by Brush Country, Black Land Prairie, Lampasas Cut Plains, Balcones Canyon Lands, Western Cross Timber, Mesquite Savannah and Live Oak-Mesquite Savannah, respectively (Table III.2).

Species richness on the 31 transects ranged from 36 to 71 species (Table A-1), with higher species richness tending to occur along an east-west band of transects lying midway between the northern and southern extremes of the study area (transect numbers 140, 28, 114, and 31, Figure III.2). Mean species richness was highest in Balcones Canyon Lands, followed by Brush Country, Mesquite Savannah with Live Oak-Mesquite Savannah, Mesquite Plains, Lampasas Cut Plains and Black Land Prairie having the same, relatively lower, richness (Table III.2).

Species diversity, based on values of H’, on the 31 transects ranged from 1.1 to 3.6 (Table A-1), with no obvious geographical gradients within the study area (Figure III.2). Mean species diversity was highest in Balcones Canyon Lands, followed by Mesquite Savannah, Lampasas Cut Plains, Live Oak-Mesquite Savannah, Black Land Prairie, Western Cross Timber, Mesquite Plains, and Brush Country (Table III.2).

Table III. 1. Mean (±1SD) avian abundance (total number of individuals of all species per 10 km of transect) and species richness (number of different species per 10 km of transect). Indices are calculated in the 8 ecological sub-regions (Figs. 1 and 2) included in the present study, with species grouped by (A) breeding habitat and (B) migration status.

A. Species grouped by breeding habitat.

	Woodland		Scrubland		Grassland		Wetland		Urban	
Ecological subregion	Avian abund.	Species rich.	Avian abund.	Species rich.	Avian abund.	Species rich.	Avian abund.	Species rich.	Avian abund.	Species rich.
Balcones Canyon Lands	20 (±5)	13 (±4)	51 (±13)	16 (±5)	3 (±2)	3 (±1)	12 (±2)	3 (±1)	45 (±23)	9 (±3)
Blackland Prairie	5 (±2)	6 (±2)	24 (±2)	7 (±1)	36 (±19)	3 (±1)	23 (±22)	5 (±2)	64 (±17)	8 (±1)
Brush Country	7 (±2)	6 (±3)	48 (±34)	14 (±2)	24 (±14)	3 (±1)	20 (±7)	5 (±1)	65 (±16)	8 (±1)
Lampasas Cut Plains	6 (±2)	7 (±1)	42 (±29)	7 (±2)	21 (±16)	3 (±1)	10 (±10)	4 (±2)	57 (±34)	8 (±2)
Live Oak-Mesquite Savannah	3 (±2)	4 (±2)	44 (±15)	15 (±2)	20 (±14)	2 (±1)	3 (±3)	2 (±1)	35 (±12)	5 (±2)
Mesquite Plains	4 (±2)	6 (±3)	39 (±21)	9 (±2)	24 (±12)	4 (±1)	12 (±5)	5 (±2)	83 (±37)	6 (±1)
Mesquite Savannah	7 (±4)	8 (±2)	41 (±15)	16 (±3)	10 (±6)	4 (±1)	7 (±4)	2 (±1)	58 (±10)	7 (±2)
Western Cross Timbers	5 (±1)	6 (±1)	38 (±1)	6 (±0)	9 (±2)	3 (±1)	4 (±1)	4 (±1)	49 (±1)	6 (±1)

Table III.1. Continued

B. Species grouped by migration status

	Neotropical migrants		Permanent residents		Short distance migrants	
Ecological subregion	Avian abundance	Species richness	Avian abundance	Species richness	Avian abundance	Species richness
Balcones Canyon Lands	47 (±19)	23 (±4)	68 (±15)	20 (±4)	50 (±14)	16 (±2)
Black Land Prairie	57 (±21)	14 (±3)	48 (±20)	11 (±1)	75 (±13)	13 (±2)
Brush Country	81 (±35)	16 (±2)	100 (±29)	17 (±3)	81 (±17)	15 (±1)
Lampasas Cut Plains	44 (±25)	16 (±3)	67 (±43)	13 (±1)	79 (±62)	14 (±3)
Live Oak-Mesquite Savannah	45 (±10)	14 (±3)	44 (±7)	13 (±1)	57 (±21)	13 (±1)
Mesquite Plains	107 (±64)	16 (±4)	68 (±34)	13 (±3)	89 (±45)	15 (±1)
Mesquite Savannah	43 (±7)	20 (±1)	62 (±1)	17 (±3)	50 (±15)	15 (±2)
Western Cross Timbers	44 (±25)	12 (±1)	59 (±0)	11 (±1)	57 (±13)	13 (±0)

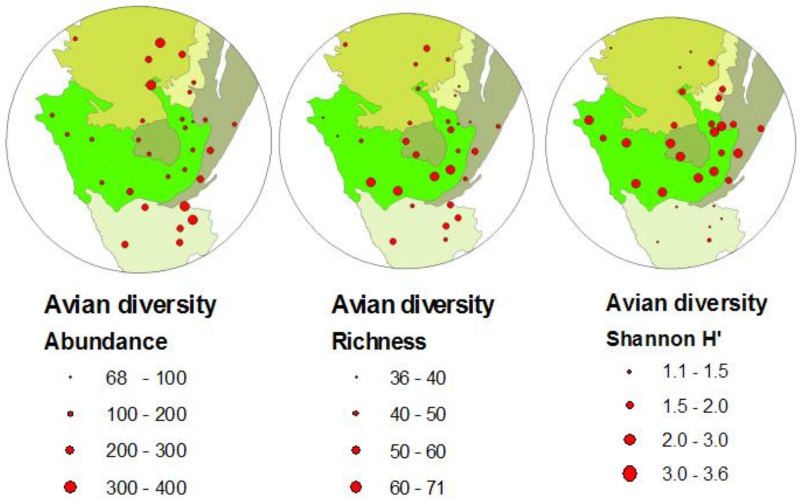

Figure III.2. *Avian abundance (total number of individuals of all species per 10 km transect), species richness (number of different species per 10 km transect), and species diversity (H').*

β-diversity, based on values of 1-S', on the 31 transects ranged from 1.2 to 2.7 (Table A-1, Figure. III.3). Mean β-diversity was lower in Brush Country, Mesquite Plains, and Balcones Canyon Lands, and higher in Live Oak-Mesquite Savannah and Lampasas Cut Plains. Highest β-diversity was founded in Live Oak Mesquite Savannah (Table III.2).

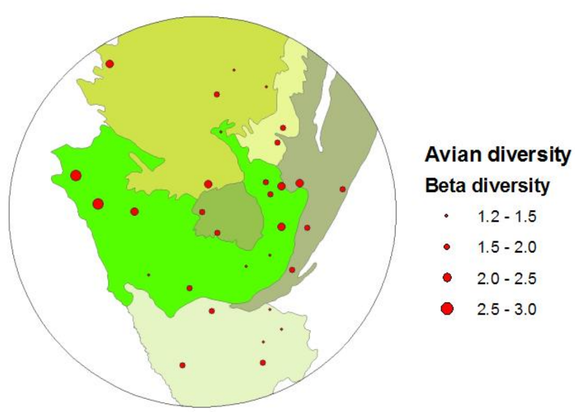

Figure III.3. β *-diversity (1-S), based on Sorensen's similarity index (S) identified in the present study.*

*Table III.2.*Mean (±1SD) avian abundance (total number of individuals of all species per 10 km of transect), species richness (number of different species per 10 km of transect), species diversity (H'), and β-diversity (1-S). Indices are calculated in the 8 ecological sub-regions (Figure III.1) included in the present study.

Ecological subregion	Abundance	Species richness	Species diversity	β-diversity
Balcones Canyon Lands	177(±27)	62(±9)	3.4 (±0.2)	1.6 (±0.3)
Black Land Prairie	200 (±56)	44 (±7)	2.9 (±0.1)	2.0 (±0.3)
Brush Country	269 (±38)	52 (±5)	1.4 (±0.2)	1.6 (±0.2)
Lampasas Cut Plains	194 (±131)	45 (±7)	3.1 (±0.1)	1.9 (±0.4)
Live Oak-Mesquite Savannah	147 (±38)	41 (±4)	3.1 (±0.1)	2.5 (±0.3)
Mesquite Plains	270 (±104)	47 (±5)	1.6 (±0.7)	1.7 (±0.5)
Mesquite Savannah	159 (±13)	52 (±4)	3.2 (±0.2)	1.9 (±0.2)
Western Cross Timbers	163 (±37)	38 (±1)	2.9 (±0.1)	1.8 (±0.0)

Landscape structure indices

Of the 10 land cover classes that resulted from aggregation of the 21 NLCD classes, 5 (woodland, scrubland, grassland, wetland, and urban) provide habitat for the avian species included in this study (Table A-4). At the landscape mosaic level, richness of patches (PR) ranged from 8 to 10, and Shannon Diversity (SHDI) ranged from 0.5 to 1.7 within the 31 buffer scenes (Table A-5).

At the land cover class level, percent of land (PL) in woodland ranged from 1 to 69, patch density (PD) from 0 to 14, mean patch size (MN) from 0 to 20, and the proximity index (PROX) from 0 to 133,109. PL in scrubland ranged from 7 to 87, PD from 1 to 26, MN from 0 to 128, and PROX from 9 to 501,314; PL in grassland ranged from 6 to 64, PD from 4 to 18, MN from 1 to 12, and PROX from 5 to 138,471; PL in wetland ranged from 0 to 1.1, PD from 0 to 2.2, MN from 0.1 to 0.7, and PROX from 0 to 35; and PL in urban ranged from 0 to 6, PD from 0 to 0.9, MN from 0 to 11, and PROX from 2 to 2,160 (Table A-5).

In the 8 ecological sub-regions, at the landscape mosaic level, Western Cross Timber, Black Land Prairie and Lampasas Cut Plains had the highest mean PR values, whereas Black Land Prairie had the highest mean value of SHDI (Table A-6). At the land cover class level, Balcones Canyon and Live Oak-Mesquite Savannah, had almost 95% of area covered by native habitats (woodland, scrubland and grassland). Balcones Canyon Lands had the highest value in

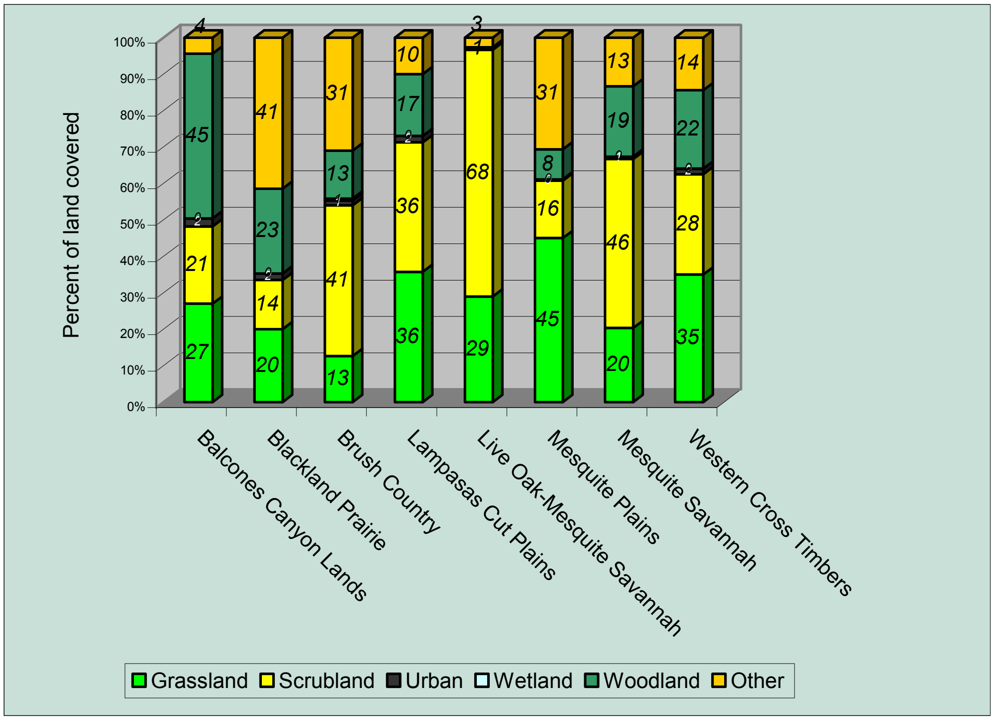

Figure III.4. *Mean percent of land in each of the 5 land cover classes in each of the 8 ecological sub-regions (Figure III.1) included in the present study.*

woodland cover, and Live Oak–Mesquite Savannah the highest in scrubland. Mesquite Savannah, Lampasas Cut Plains and Western Cross Timber, had almost 85% of area covered in native habitats, especially in scrubland and grassland. Black Land Prairie, Brush Country and Mesquite Plains, had less values in area covered by natural habitats, around 65% (Table A-6, Figure III.4).

Balcones Canyon Lands were characterized by woodlands (PL = 45), with low PD (7/100 ha), small MN (7,8 ha), and high values of PROX (50323), indicating relatively little fragmentation of woodlands (Table A-6, Figures III.4, III.5A and III.6A).

Live Oak-Mesquite Savannah and Brush Country both were characterized by scrubland, but Live Oak-Mesquite Savannah offers the better quality of habitat. Live Oak-Mesquite Savannah was more than half covered by scrubland (PL = 67.7), with big patches MN (48.7 ha) which are in close proximity to each other, as indicated by a PROX (301,948) (Table A-6, Figures III.4, III.5B and III.6B).

Mesquite Plains and Lampasas Cut Plains were characterized by grassland (PL = 45 and 35.8, respectively), but the former had the less-fragmented habitat, MN (7.8) and PROX (68458) compared with Lampasas Cut Plains, which presented MN(4.5) and PROX(3155) (Table A-6, Figures III.4, III.5C, and III.6C).

Balcones Canyon Lands, Lampasas Cut Plains, Black Land Prairie and Brush Country were the most urbanized sub-regions (PL = 2.2, 1.8, 1.8, and 1.3, respectively). Mean patch size (MN) was 5, 6.5, 4.3, and 3.3 ha, respectively. Proximity index (PROX) was highest in Brush Country (373), followed by Balcones Canyon Lands (281), Lampasas Cut Plains (174), and Black Land Prairie (76), indicating that urban zones were more compact in Brush Country, Balcones Canyon Lands, and Lampasas Cut Plains (Table A-6, Figures III.4, III.5D, and III.6D).

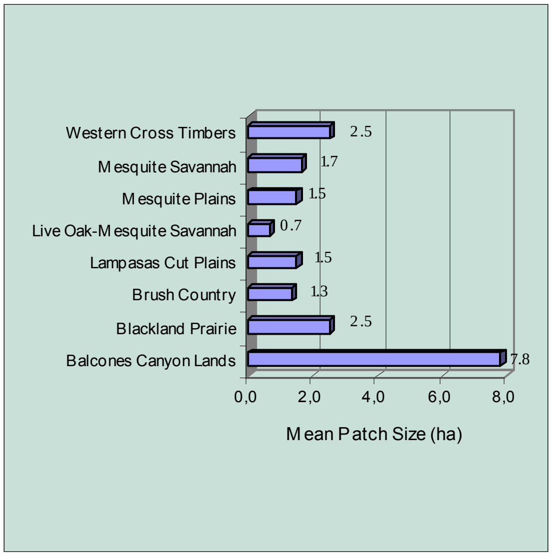

Figure III.5A. Woodland

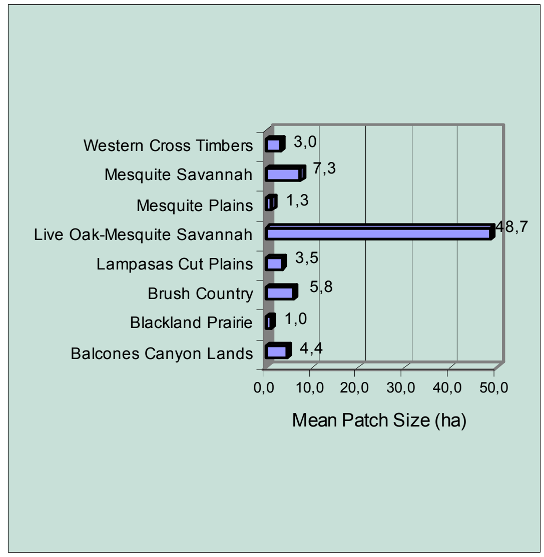

Figure III.5B. Scrubland.

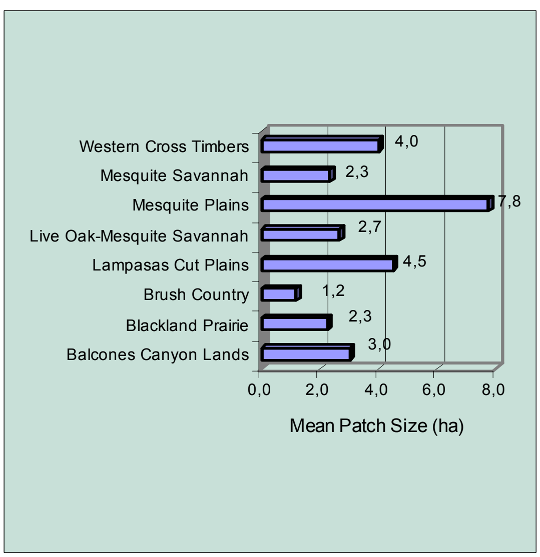

Figure III.5C Grassland.

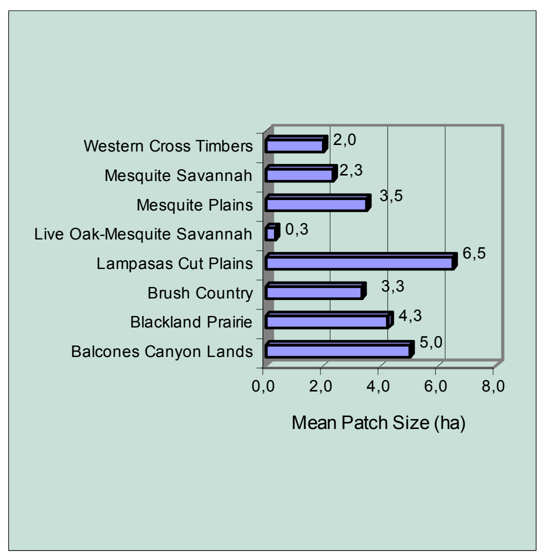

Figure III.5D. Urban

Figure III.5. *Mean patch size (MN) in the 8 ecological subregions*

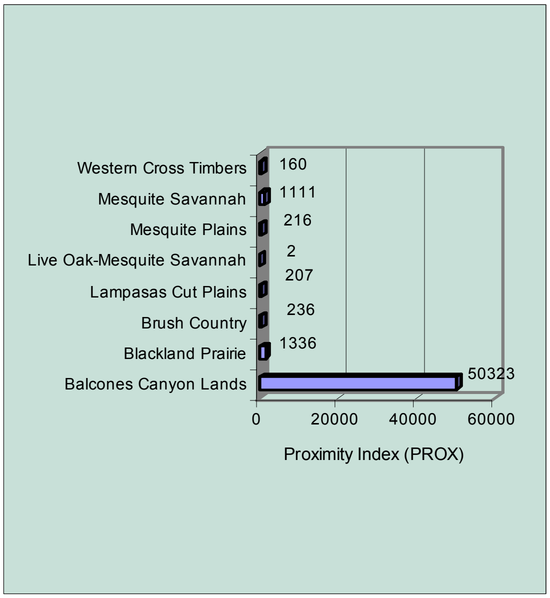

Figure *III.6*A. Woodland

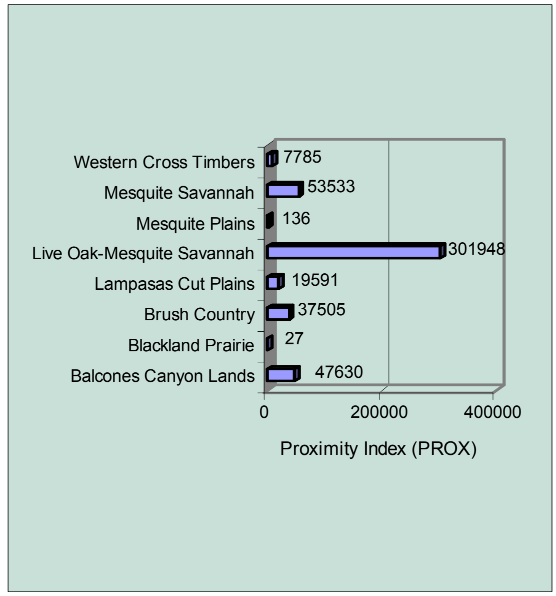

Figure *III.6*B. Scrubland

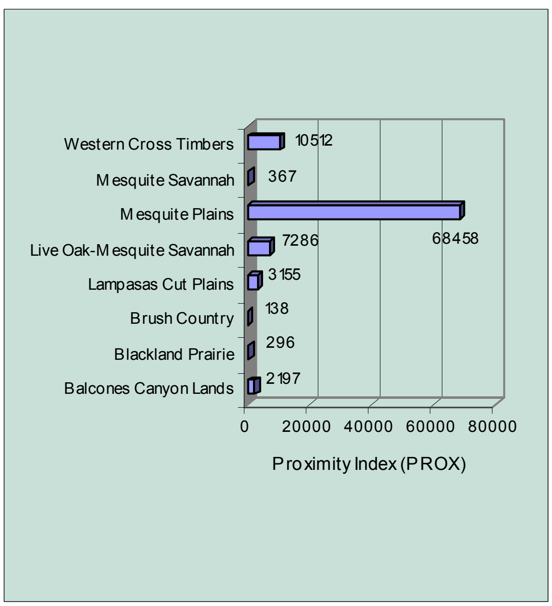

Figure *III.6*C. Grassland

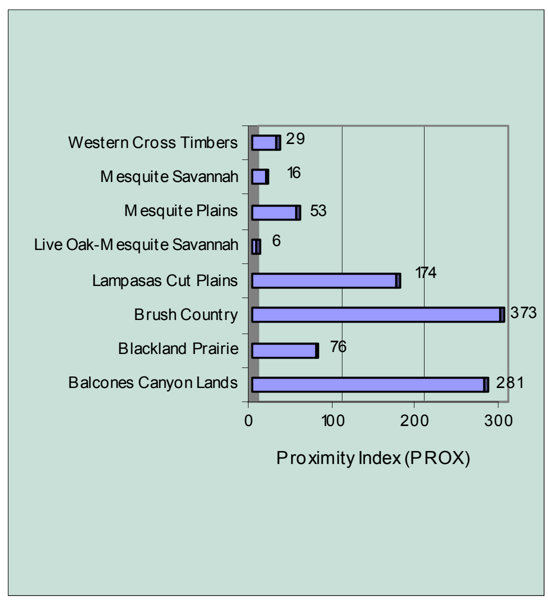

Figure *III.6*D. Urban

Figure III.6. *Proximity index of land cover classes (A) woodland, (B) scrubland, (C) grassland and (D) urban, in the 8 ecological subregions.*

Spatial autocorrelation of variables.

Approximately half of the indexes showed significant levels of spatial autocorrelation, including avian abundance, H', and (1-S) (P = 0.0531, 0.0015, and 0.0141, respectively), but not species richness (P = 0.4531), among avian diversity indexes, and SHDI (P = 0.0072), but not PR (P = 0.2406), among landscape structure indexes at the landscape mosaic level (Table III.3). Landscape structure indexes at the land cover class level (PL, PD, MN, and PROX) tended to be more significantly spatially autocorrelated (P < 0.1) in scrublands, grasslands, and wetlands than in woodlands and urban areas (P > 0.1), although MN was not significantly autocorrelated in scrublands (P = 0.2952) and PD was significantly autocorrelated in woodlands (P = 0.0001).

Spatial correlations between pairs of avian diversity and landscape structure indexes

Avian abundance, species richness, and H' were not significantly spatially correlated with landscape structure indexes at the landscape mosaic level (PR and SHDI) (P > 0.28); (1-S) was not significantly spatially correlated with PR (P = 0.1220) but did show a significant positive spatial correlation with SHDI (P = 0.0253) (Table III.4).

At the land cover class level, both avian abundance and species richness showed a significant positive spatial correlation with PL, MN, and PROX (P < 0.052), but no significant spatial correlation with PD (P > 0.23), in woodlands, and species richness showed a significant positive spatial correlation with PL, PD, and PROX (P < 0.041), but no significant spatial correlation with MN (P = 0.0967), in scrublands.

There were no significant spatial correlations (P > 0.11) between any pairs of avian diversity and landscape structure indexes at the land cover class level in grasslands, wetlands, or urban areas.

Table III.3. Degree of spatial autocorrelation. Calculation is made for each of 4 indexes of avian diversity (avian abundance, species richness, species diversity (H'), β-diversity (1-S)) and each of 6 indexes of landscape structure (2 at the landscape mosaic level (richness of patches (PR) and Shannon Diversity (SHDI)) and 4 at the land cover class level (percent of land (PL), patch density (PD), mean patch size (MN), proximity (PROX))), as indicated by Mantel's r (Fortin et al., 1993).

	Indexes	Mantel's r	P
Avian Diversity			
	Avian Abundance	0.1427	0.0531
	Species Richness	0.0051	0.4531
	H'	0.2632	0.0015
	(1-S)	0.2012	0.0141
Landscape Structure			
Landscape Mosaic Level	PR	0.0498	0.2406
	SHDI	0.2643	0.0072
Land Cover Class Level			
Woodland	PL	0.1009	0.1517
	PD	0.4162	0.0001
	MN	-0.0248	0.5400
	PROX	-0.1049	0.8365
Scrubland	PL	0.2216	0.0071
	PD	0.1303	0.0521
	MN	0.0570	0.2952
	PROX	0.1726	0.0884
Grassland	PL	0.3811	0.0001
	PD	0.1742	0.0416
	MN	0.3734	0.0005
	PROX	0.3140	0.0070
Wetland	PL	0.2164	0.0295
	PD	0.1861	0.0603
	MN	0.2970	0.0063
	PROX	0.1889	0.0605
Urban	PL	-0.0770	0.7763
	PD	0.0696	0.1807
	MN	-0.0643	0.7308
	PROX	-0.0859	0.7581

Table III.4. Degree of spatial correlation. Calculation is made between pairs of avian diversity (avian abundance, species richness, species diversity (H'), β-diversity (1-S)) and landscape structure ((richness of patches (PR), Shannon Diversity (SHDI), percent of land (PL), patch density (PD), mean patch size (MN), proximity (PROX)) indexes, as indicated by Cross Mantel's r (Fortin et al., 1993).

		Avian Diversity							
Landscape Structure	Indexes	Avian Abundance Cross Mantel's r	P	Species Richness Cross Mantel's r	P	H' Cross Mantel's r	P	(1-S) Cross Mantel's r	P
Landscape Mosaic Level	*PR*	-0.0095	0.4839	-0.0165	0.5341	-0.0862	0.9736	0.0935	0.1220
	SHDI	0.0033	0.4291	0.0382	0.2831	-0.0236	0.5630	0.2255	0.0253
Land Cover Class Level									
Woodland	PL	0.6042	0.0002	0.4098	0.0025				
	PD	0.0080	0.4143	0.0621	0.2346				
	MN	0.5058	0.0059	0.3185	0.0514				
	PROX	0.6157	0.0027	0.4216	0.0100				
Scrubland	PL	0.1003	0.1006	0.2199	0.0099				
	PD	0.0009	0.3957	0.1627	0.0289				
	MN	0.0445	0.1775	0.0760	0.0967				
	PROX	0.0981	0.1580	0.2073	0.0407				
Grassland	PL	-0.0112	0.4835	0.0345	0.2773				
	PD	-0.0029	0.4408	0.1081	0.1536				
	MN	-0.0536	0.6900	0.0720	0.2210				
	PROX	-0.0323	0.5132	0.1023	0.1957				
Wetland	PL	-0.0036	0.3284	0.0209	0.3683				
	PD	-0.0301	0.4505	-0.0178	0.4959				
	MN	-0.0618	0.6066	0.0567	0.2681				
	PROX	-0.0435	0.5040	0.0359	0.3025				
Urban	PL	-0.0407	0.5917	0.1589	0.1136				
	PD	0.0043	0.4312	-0.0363	0.5887				
	MN	0.0215	0.3579	0.1096	0.1369				
	PROX	-0.0700	0.6222	0.0683	0.2282				

Correlation between values of pairs of avian diversity and landscape structure indexes

After appropriate adjustments for spatial autocorrelation, there were no significant correlations between values of pairs of avian diversity (avian abundance, species richness, H', (1-S)) and landscape structure indexes at the landscape mosaic level (PR, SHDI) (P > 0.068) (Table III.5). At the land cover class level, both avian abundance and species richness showed a significant positive correlation with PL, MN, and PROX (P < 0.017), but no significant correlation with PD (P > 0.42), in woodlands, and species richness showed a significant negative correlation with PD (P = 0.0194) in grasslands. There were no significant correlations (P > 0.08) between any pairs of avian diversity and landscape structure indexes at the land cover class level in scrublands, wetlands, or urban areas.

Table III.5. Degree of correlation. Calculation is made between values of pairs of avian diversity (avian abundance, species richness, species diversity (H'), β-diversity (1-S)) and landscape structure ((richness of patches (PR), Shannon Diversity (SHDI), percent of land (PL), patch density (PD), mean patch size (MN), proximity (PROX))) indexes, as indicated by the Modified t-Test for autocorrelation (CHR) (Clifford et al., 1989; Dutilleul, 1993).

		Avian Diversity							
		Avian Abundance		Species Richness		H'		(1-S)	
Landscape Structure	Indexes	CHR	P	CHR	P	CHR	P	CHR	P
Landscape Mosaic Level	*PR*	0.1413	0.4592	-0.1446	0.3972	0.0577	0.7493	-0.0360	0.8387
	SHDI	0.3785	0.0682	-0.1036	0.7130	-0.3159	0.2763	-0.3829	0.1096
Land Cover Class Level									
Woodland	PL	0.7048	0.0012	0.5987	0.0059				
	PD	0.0761	0.7648	0.1735	0.4248				
	MN	0.6281	0.0009	0.4918	0.0166				
	PROX	0.7050	0.0022	0.6066	0.0035				
Scrubland	PL	0.3474	0.1314	0.4892	0.1399				
	PD	-0.3589	0.1322	-0.4586	0.1592				
	MN	0.1994	0.3378	0.2820	0.2635				
	PROX)	0.3854	0.0801	0.4615	0.1306				
Grassland	PL	-0.0365	0.8345	0.1175	0.5095				
	PD	0.1518	0.4407	-0.4780	0.0194				
	MN	-0.0336	0.8297	0.2662	0.1545				
	PROX	-0.2037	0.1735	-0.0202	0.8991				
Wetland	PL	0.2003	0.4165	0.3112	0.2124				
	PD	0.1969	0.4142	0.3023	0.1894				
	MN	0.0860	0.5976	0.3491	0.1953				
	PROX	0.1184	0.4572	0.2913	0.2035				
Urban	PL	0.0494	0.8401	0.3947	0.1191				
	PD	0.0617	0.7456	-0.0251	0.9069				
	MN	-0.0733	0.7588	0.2947	0.2066				
	PROX	0.0266	0.8816	0.2826	0.1339				

Discussion

Avian diversity (α-diversity and β-diversity) and landscape structure were correlated spatially within the Edwards Plateau Ecoregion of Texas. Especially important for conservation planning, are spatial correlations between β-diversity and SHDI at the landscape mosaic level, and the spatial correlations between α-diversity (abundance and richness) and mean patch size (MN) and α-diversity and proximity (PROX) at the land cover class level. The former is important for western portion of the Edwards Plateau in the Live Oak-Mesquite Savannah, while the latter is important for eastern portion, the Balcones Canyon Lands sub-region.

At the landscape mosaic level, positive significant spatial correlation between β-diversity and SHDI (Table III.4) were found, using the Cross Mantel test, meaning that β-diversity is positively associated with habitat diversity. The Live Oak-Mesquite Savannah sub-region, presented the highest β-diversity values (Table III.2), indicating that in this sub-region, there are high habitat richness and biological differences between patches of different habitats (Mumby 2001) that produce an interchange of species resulting in different bird communities.

Coincidently, 97% of the area of this ecological subregion was covered by natural habitat class cover as scrubland and grassland (68% and 29% respectively), and presented high values of landscape richness PR (9.3), (Table A-6, Figure III.4). These results indicate that in this ecological subregion it is important to conserve patch richness – e.g., it cannot be converted in a homogeneous landscape of pastures because bird turnover needs landscape diversity.

At the class level, there were positive high significant correlations resulting with both the cross Mantel (Table III.4) and the Modified *t*-test (Table III.5) between α-diversity (richness) and class level (mean patch size MN and proximity index PROX). These correlations are important in Balcones Canyon Lands ecological sub-region, where highest values of woodland and scrubland bird richness (Table III.1) occurred, and where highest values of richness and H' (Table III.2) were observed.

This ecological subregion also had a high percentage of land covered by natural habitats (93%), represented by woodland (45%), grassland (27%) and scrubland (21%), Figure III.4. Woodland cover presented the highest values of mean patch size MN (7,8 ha) and PROX (50323), indicating relatively little fragmentation of woodlands, Table A-6; Figures III.5A and III.6A.

However, the geographic location of this ecological subregion (the eastern border of the Edwards Plateau), and the fact it is one of most urbanized ecological subregions (PL =2.2, MN=5, and PROX=281), make woodland habitat vulnerable to fragmentation, due to the urban growth of the Austin-San Antonio-San Marcos Metropolitan belt. This expansion will probably produce a decrease in patch size (MN) and an increase in isolation, producing a negative impact on avian

α-diversity. Population sizes depend on the sizes of and distances among habitat patches. In addition, α-diversity depends on the suitability of the increasing urban habitat separating the favorable patches. So colonization of patches to maintain native bird populations can be difficult and can produce local extinctions of native woodland and scrubland species.

Balcones Canyon Lands had the highest number of urban bird richness (9), Table III.1A, which can be an indication of replacement of natural habitats as woodland and scrubland, by urban zones. On the other hand, 27% of neotropical migrants and 23% of the permanent residents of the ecorregion, were registered in this ecological subregion, so it is a priority to manage the environmental conditions that support highest richness of main habitats in Balcones Canyon Lands sub-region.

Positive spatial correlations found between species richness and scrubland class structure (Pl, PD, and PROX), indicating that spatial arrangement, size and proximity of patches, are important for conservation in the in Live Oak-Mesquite Savannah. The scrubland-grassland scatter landscape with high number of big scrubland patches MN (48.7 ha) in close proximity to each other, PROX (301,948), mixed with small grassland patches (2,7 ha), can be the key to preserve the appropriate environmental conditions for the 15 scrubland bird species and the 2 grassland bird species that were registered in the Live Oak-Mesquite Savannah (Tables A-6 and III.1A).

Conclusions

Avian diversity (α-diversity and β-diversity) and landscape structure were correlated spatially within the Edwards Plateau Ecoregion of Texas. At the landscape mosaic level, positive significant spatial correlation between β-diversity with SHDI and PR (Table III.4) showed the importance of preserving all types of land cover (maximum richness) and configuration, in order to preserve the β-diversity .

At the class level, highly significant positive correlations, between α-diversity (richness) and class level (mean patch size MN and proximity PROX), showed the importance of maintaining mean patch size around 8 ha for woodland and 45 ha in scrubland, and a minimum distance intermediate to that used by permanent resident species and short-distance migrants in breeding movements, in order to preserve the woodland and scrubland bird species richness.

For conservation planning, positive spatial correlation at land mosaic level is important for management of Live Ok-Mesquite Savannah, because this sub-ecoregion has high values of β-diversity. Positive spatial correlations at the class level are important to management of Balcones Canyon Lands because it is the sub-ecoregion with high values of α-diversity (richness and Shannon H').

CHAPTER IV

SPATIAL RELATIONSHIPS BETWEEN LANDSCAPE STRUCTURE AND OWNERSHIP PROPERTY SIZE IN THE EDWARDS PLATEAU OF TEXAS: IMPLICATIONS FOR CONSERVATION PLANNING

Introduction

Human-induced landscape transformations have important implications for the maintenance of biodiversity. Ecological processes are related not only to land use, but also to landscape structure, that is, to the spatial arrangement of land elements (Zonneveld and Forman 1989; Baudry 1993). Among the more pervasive landscape changes whose impact on biodiversity is potentially great, yet remains poorly understood, is that associated with changes in ownership property sizes along urban – rural gradients. Urban sprawl generates economic pressure extending well past city limits into the rural landscape (Costanza et al.1997), which leads to a reduction in ownership property sizes (Adger and Luttrel 2000; Antrop 2000; Swenson and Franklin 2000; Luck and Wu 2002). Reduction in property sizes leads to changes in landscape structure (Stanfield et al. 2002; Wilkins et al. 2003), which can lead to changes in biodiversity (Donovan and Flather 2002; Lovett-Doust et al. 2003).

In Texas, the fragmentation of large, family-owned farms and ranches has been identified as the greatest threat to wildlife habitat within the state (Shackelford and Shackelford 2003; Wilkins et al. 2003). Many rural areas are experiencing greatly increased residential development; the southern and western portions of the state (the Trans Pecos, Edwards Plateau, South Texas Brush Country, and Coastal Sand Plains ecoregions) have been loosing annually more than 235,000 acres that were in large ownerships (>2000 acres), thus dramatically shifting the size-class distribution of farms within these regions (Wilkins et al. 2003). In a case study within a single county (Bastrop), Wilkins et al. (2003) found that landscape characteristics such as number of patches per unit area and average patch size of native rangeland were influenced significantly by the subdivision of farms and ranches, that is, by land ownership fragmentation. I am unaware of other studies within Texas that quantitatively relate the distribution of land ownership sizes to specific landscape characteristics.

As the fragmentation of rural properties continues, conservation planning must be based on a more thorough understanding of the linkages between ownership property sizes and specific elements of landscape structure. We now have both the conceptual framework of landscape ecology (Forman 1984; Zonneveld and Forman 1989; Turner 1989; McGarigal 1995; Turner et al. 2001; Gergel and Turner 2002) and the quantitative tools of spatial statistics (Isaaks and Srivastava 1989; Legendre and Fortin 1989; Fortin and Gurevitch 1993; Dutilleu et al. 1993; Dale and Fortin 2002; Fortin and Payette 2002) with which to do so. In this chapter, I investigate specific linkages between ownership property size and landscape structure within the Edwards Plateau of Texas. I first test for statistically significant spatial correlations among 6 indexes of landscape structure and 4 ownership property sizes, and then explore the implications of these results for conservation planning.

Study area

The study area was bounded by a circle of radius 300 km centered at the geographical center point of the Edwards Plateau Ecoregion, which includes all of the Edwards Plateau and portions of 5 other ecoregions--South Texas Brush, Blackland Prairie, Llano Uplift, Rolling Plains, and Oak Woods. For purposes of the present study, I used the 8 Texas ecological subregions (Gould 1975. adapted by Wu et al. 2002), included in these ecoregions: Balcones Canyon Lands, Black Land Prairies, Brush Country, Lampasas Cut Plains, Live Oak-Mesquite Savannah, Mesquite Savannah, Mesquite Plains, and Western Cross Timbers (Figure IV.1).

The climate ranges from subtropical steppe to subtropical sub-humid, with mean annual precipitation ranging from 375 mm in the west to 750 mm in the east, about three-fourths of which falls during the growing season (April through mid-November). The area is predominantly shrub land grazed by cattle, sheep, and goats, but local tracts are cultivated for domestic pasture and hay; cotton and grain sorghum are grown locally on irrigated land and there are some pecan orchards on flood plains. Landowners commonly lease their land for hunting deer, quail, mourning dove, wild turkey, and/or javelina. Many rural areas are experiencing greatly increased residential development, especially in the eastern portion of the region, due in large part to the influence of large cities such as San Antonio and Austin (Wilkins et al. 2003).

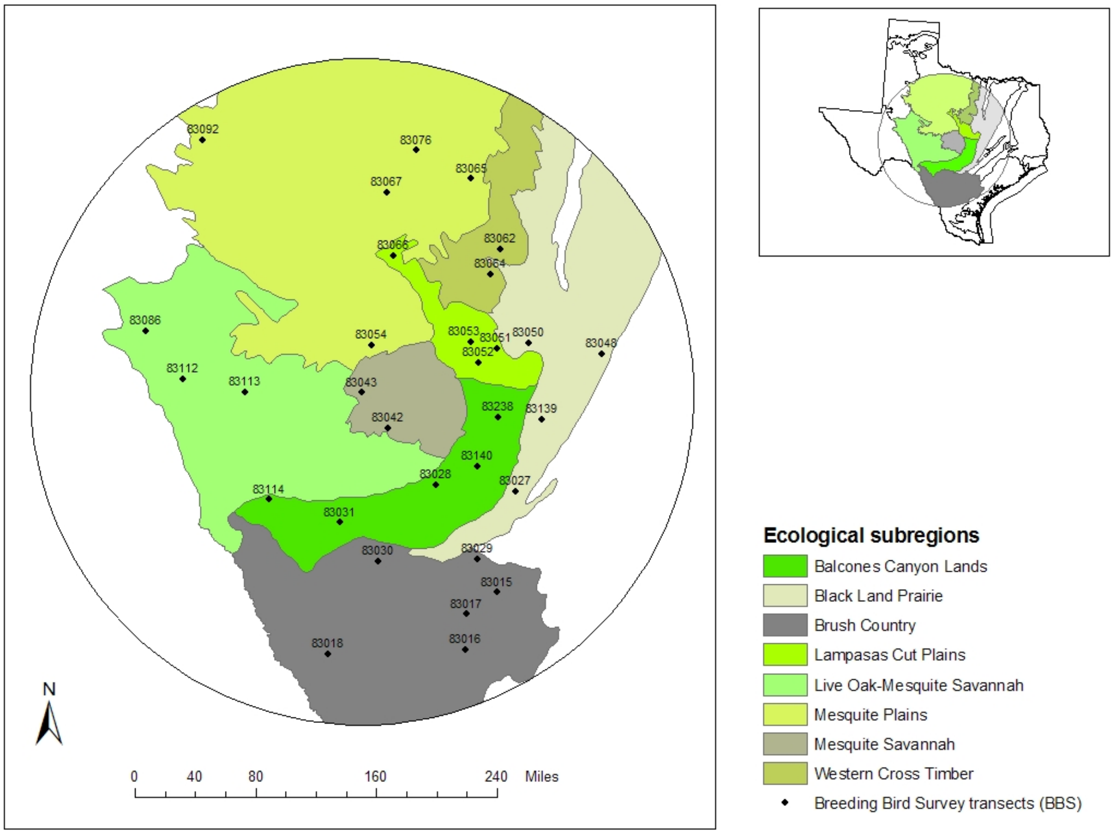

Figure IV.1. *Geographic location of the study area. The study area included the 8 ecological sub-regions referred to in this dissertation. Locations of the 31 North American Breeding Bird Survey (BBS) transects included in the present study also are indicated; numbers correspond to the BBS code, of which the first two digits (83) signify Texas, and the final three digits (015-238) correspond to the transect number.*

Methods

I first describe the databases and the calculation of the indexes used to represent ownership property size and landscape structure. Second, I describe the test for spatial autocorrelation for each variable (index). I then describe the tests for spatial correlations between each pair of variables. Finally, I describe the test for correlation between the values of each pair of variables, corrected for effects of spatial correlation.

Ownership property sizes

I obtained data on ownership property sizes in 1992 for each of the 27 Texas counties that contained one or more of the 31 North American Breeding Bird Survey (BBS) transects described in Chapter II. These data included average size of rural (farm and ranch) property in acres (USDA. 1992) and the proportions of rural acreage in each of 4 size classes: <50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres (unpublished data, Land Information Systems Laboratory, Texas A&M University; data summary available in Wilkins, Hays, Kubenka, Steinbach, Grant, González, Kjelland, and Shakelford., 2003). I also calculated the mean and standard deviation of the proportions of acreage in each of the 4 size classes in each of the 5 ecological sub-regions.

Landscape structure indexes

I obtained land cover data for the Edwards Plateau Ecoregion for 1992 from the National Land Cover Dataset (NLCD) of the United States Geological Service (USGS); more recent data were not available. The spatial resolution of the data is 30 meters, mapped in the Albers Conic Equal Area projection, NAD 83. I first regrouped the 21-class land cover classification scheme of the NLCD into 10 land cover classes, using the Spatial Analyst tool of Arc 8 GIS (ESRI 2000). I then identified which of these more aggregated land cover classes provide habitat for the avian species seen on the BBS transects included in this study.

I then created, using the Buffer Wizard tool in Arc 8 GIS (ESRI 2000), buffer scenes of 5, 10, and 20 km around the 31 BBS transects (buffer shape files). I used each buffer shape file to “cut” an identical area on the NLCD using the Spatial Analyst tool; I used these scenes to relate landscape structure to avian diversity (Donovan and Flather 2002).

I then calculated, within each of the 31 buffer scenes, 6 indexes of landscape structure, 2 at the landscape mosaic level, and 4 at the land cover class level. At the landscape mosaic level, I calculated richness of patches (PR), which is the number of types of land cover classes present in the landscape of each buffer scene, excluding the landscape border if present, and Shannon Diversity (SHDI):

$$SHDI = -\sum_{i=1}^{m} P_i * \ln P_i \qquad \text{(IV.1)}$$

where P_i is the proportion of the landscape of each buffer scene occupied by land cover class i.

At the land cover class level, I calculated for each of the land cover classes, percent of land PL, patch density PD (number of patches / 100 ha), mean patch size MN (ha), and the proximity index (PROX), using FRAGSTATS 2.0 (McGarigal and Marks 1995; Stanfield et al. 2002).

$$PROX = \sum_{s=1}^{n} \frac{a_{ijs}}{h_{ijs}^{2}} \qquad \text{(IV.2)}$$

where PROX represents the proximity index for focal patch i, a_{ijs} is the area (m^2) of patch ijs within a specified neighborhood (m) of patch ij, and h_{ijs} is the distance (m) between patch ijs and patch ijs, based on patch edge-to-edge distance, computed from cell center to cell center. I selected a 100 meter search radius under the assumption that this was within the daily range of movements of all of the avian species included in this study. Low index values indicate patches that are relatively isolated from other patches within the specified buffer distance, and high values indicate patches that are relatively connected to other patches (Turner et al. 2001).

I used PROX as an indicator of wildlife habitat fragmentation; high values indicate less fragmentation and low values indicate more fragmentation (Mortberg 2001; Brooks et al. 2002). I also calculated the mean and standard deviation of PR, SHDI, PL, PD, MN, and PROX of the buffer scenes in each of the 5 ecological sub-regions.

Spatial autocorrelation of variables

I tested for spatial autocorrelation in each of the ownership property sizes and each of the indexes of landscape structure using a Mantel Tests (Fortin and Gurevitch 1993). The Mantel Test (r) is a regression in which the variables themselves are distance, or dissimilarity (ecological distances), matrixes summarizing pair-wise similarities among sample locations:

$$r = \frac{\sum\sum stdA_{ij} * stdB_{ij}}{n-1} \quad \text{(sum from i to n and sum from j to n, for i} \neq \text{j)} \qquad \text{(IV.3)}$$

where n is the number of sample locations, i and j identify the matrix element, B_{ij} is the Euclidian distance matrix of location points, and A_{ij} is the dissimilarity matrix of the variable of interest, the present case, S1, S2, S3, S4, PR, SHDI, PL, PD, MN, or PROX. For Mantel test calculations, I used the PASSAGE program (Rosenberg 2005).

Spatial correlations between pairs of ownership property sizes and landscape structure indexes

To determine if ownership property sizes and the indexes of landscape structure are spatially correlated, I conducted a Cross Mantel Test (r) (Fortin and Gurevitch 1993), between each pair of ownership property sizes and landscape structure indexes.

This set of analyses tests for a spatial relationship *per se* between differences of pairs of values of both variables (ownership property sizes and the indexes of landscape structure), but does not indicate the degree of correlation between the values of the two variables. For example, if the differences in the values of variable A and the differences in the values of variable C increase or decrease with increasing distance (that is, with increasing geographical distance between the points where the values of the variables were measured), the variables A and C are positively spatially correlated. If the differences in the values of variable A increase with distance and the values of variable C decrease with distance, or vice versa, then variables A and C are negatively spatially correlated.

$$r = \frac{\sum \sum stdA_{ij} * stdC_{ij}}{n-1}$$ (sum from i to n and sum from j to n, for i ≠ j) (IV.4)

where n is the number of sample locations, i and j identify the matrix element, A_{ij} is the dissimilarity matrix of one of the variables of interest, that is an index of landscape structure (PR, SHDI, PL, PD, MN, PROX), and C_{ij} is the dissimilarity matrix of the other variable of interest, that is, an ownership property size (S1, S2, S3, S4).

Correlation between values of pairs of ownership property sizes and landscape structure indexes

I conducted a Pearson's pair-wise correlation (ρ) between each pair of ownership property sizes and landscape structure indexes. This set of analyses identifies the degree of correlation between the values of the two variables,

$$\rho = \frac{\sum_{i=1} (u_i - m_u)(v_i - m_v)}{S_u S_v}$$ (IV.5)

where u and v are two variables (u is one of the 6 landscape structure indexes and v is one of the 4 ownership property sizes),. m_u and m_v are their respective means, and S_u and S_v are their respective standard deviations. I then used the Modified *t*-test for autocorrelation (CHR), which corrects the degrees of freedom based on the amount of autocorrelation in the data, to assess the correlation between each pair of spatially correlated variables (Clifford et al. 1989; Dutilleul 1993). The procedure calculates the amount of spatial autocorrelation of variables to determine how different the effective sample size (n') is from the number of observations.

$$n'(R) = \frac{n^2}{\sum\sum cor(u_i u_j)} \qquad \text{(IV.6)}$$

where R is the autocorrelation matrix, n is the number of observations, and u_i, and u_j are the observations of the two variables. The corrected degrees of freedom (n'-2) were then used to test the significance of the correlation; I used the PASSAGE program (Rosenberg, M. 2005) to perform these calculations.

Results

Ownership property sizes

Mean ownership property sizes for counties ranged from 276 acres (Caldwell County) to 12,746 acres (Crocket County) (Table A-7), increasing along an east-west gradient (Figure. IV.2). Largest average of ownership property sizes by ecological subregion, occurred in Live Oak-Mesquite Savannah, followed by Mesquite Plains and Balcones Canyon Lands, Mesquite Savannah, Brush Country, Lampasas Cut Plains, and Western Cross Timber , with maximum and minimum sizes, and ranges between maximum and minimum sizes, all decreasing in generally the same order, with a few exceptions (Table IV.1).

Although a relatively high percentage of the rural acreage was in the largest (S4, >500 acres; from 16 to 75%) and next-to-largest (S3, 100-500 acres; from 14 to 45%) size classes in all 8 ecological sub-regions, 7 of the 8 sub-regions also had roughly one-fifth of the rural acreage in the smallest (S1, <50 acres; from 7 to 25%) size class, the exception being Live Oak-Mesquite Savannah (only 7% in S5) (Figure. IV.3).

Table IV.1. *Mean, maximum, and minimum ownership property sizes (OPS) in acres, the range of sizes (maximum - minimum). Calculation is made for counties in each of the 8 ecological sub-regions included in this study.*

Ecological subregion	Data	Mean OPS	S1	S2	S3	S4
Balcones Canyon Lands	Mean	1441	20	10	34	36
	Max	4232	28	17	41	69
	Min	298	8	2	21	17
Blackland Prairie	Mean	387	20	16	45	20
	Max	609	28	19	52	28
	Min	276	12	9	38	14
Brush Country	Mean	965	17	12	43	28
	Max	3288	25	18	55	55
	Min	281	10	6	29	11
Lampasas Cut Plains	Mean	625	17	10	46	29
	Max	636	22	11	52	31
	Min	609	12	9	42	25
Live Oak-Mesquite Savannah	Mean	7305	7	4	14	75
	Max	12746	13	6	19	90
	Min	2964	3	2	4	66
Mesquite Plains	Mean	1530	13	9	39	40
	Max	2391	22	13	46	56
	Min	706	8	6	30	26
Mesquite Savannah	Mean	1089	9	4	37	50
	Max	1278	9	4	41	52
	Min	995	9	4	35	46
Western Cross Timbers	Mean	349	25	17	42	16
	Max	355	29	19	47	18
	Min	343	21	14	37	14

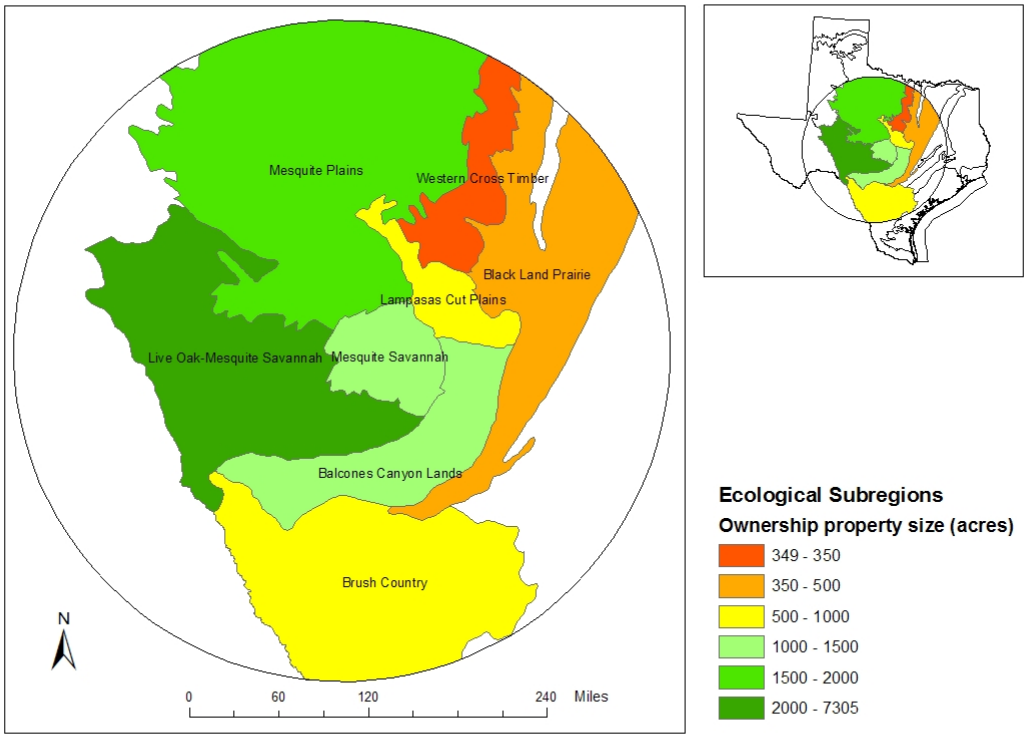

Figure IV.2. *Ownership property size in the ecological subregions.*

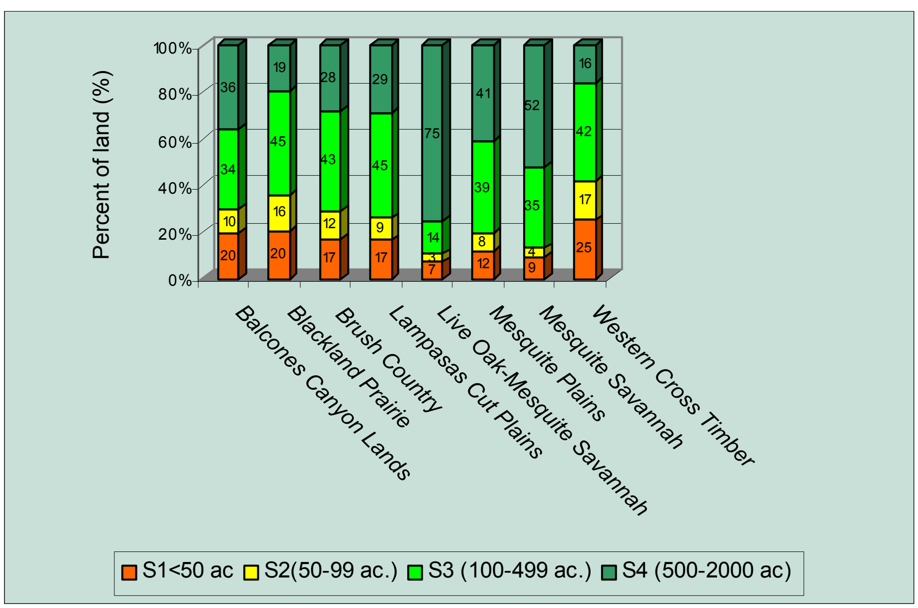

Figure IV.3. *Mean percent of rural acreage in each of 4 size classes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres). Data is given for counties within each of the 8 ecological sub-regions included in this study.*

Landscape structure indices

Of the 10 land cover classes that resulted from aggregation of the 21 NLCD classes, 5 (woodland, scrubland, grassland, wetland, and urban) provide habitat for the avian species included in this study (Table A-4). At the landscape mosaic level, richness of patches (PR) ranged from 8 to 10, and Shannon Diversity (SHDI) ranged from 0.5 to 1.7 within the 31 buffer scenes (Table A-5).

At the land cover class level, percent of land (PL) in woodland ranged from 1 to 69, patch density (PD) from 0 to 14, mean patch size (MN) from 0 to 20, and the proximity index (PROX) from 0 to 133,109. PL in scrubland ranged from 7 to 87, PD from 1 to 26, MN from 0 to 128, and PROX from 9 to 501,314; PL in grassland ranged from 6 to 64, PD from 4 to 18, MN from 1 to 12, and PROX from 5 to 138,471; PL in wetland ranged from 0 to 1.1, PD from 0 to 2.2, MN from 0.1 to 0.7, and PROX from 0 to 35; and PL in urban ranged from 0 to 6, PD from 0 to 0.9, MN from 0 to 11, and PROX from 2 to 2,160 (Table A-5).

In the 8 ecological sub-regions, at the landscape mosaic level, Western Cross Timber, Black Land Prairie and Lampasas Cut Plains had the highest mean PR values, whereas Black Land Prairie had the highest mean value of SHDI (Table A-6). At the land cover class level, Balcones Canyon and Live Oak-Mesquite Savannah had almost 95% of area covered by native habitats (woodland, scrubland and grassland). Balcones Canyon Lands had the highest value in woodland cover, and Live Oak–Mesquite Savannah the highest in scrubland. Mesquite Savannah, Lampasas Cut Plains and Western Cross Timber, had almost 85% of area covered in native habitats, especially in scrubland and grassland. Black Land Prairie, Brush Country and Mesquite Plains, had less values in area covered by natural habitats, around 65% (Table A-6, Figure IV.4).

Balcones Canyon Lands were characterized by woodlands (PL = 45), with low PD (7/100 ha), small MN (7,8 ha), and high values of PROX (50323), indicating relatively little fragmentation of woodlands (Table A-6, Figures IV.4, IV.5A and IV.6A).

Live Oak-Mesquite Savannah and Brush Country both were characterized by scrubland, but the former has the best quality of habitat. Live Oak-Mesquite Savannah was more than half covered by scrubland (PL = 67.7), with big patches MN (48.7 ha) which are in close proximity to each other, as indicated by a PROX (301,948), Table A-6 (Figures 6, 7B, 8B Chapter III). Mesquite Plains and Lampasas Cut Plains were characterized by grassland (PL = 45 and 35.8, respectively), but first one had the habitat less fragmented, MN (7.8) and PROX (68458) compared with Lampasas Cut Plains, which presented MN(4.5) and PROX(3155) (Table A-6, Figures IV.4, IV.5C, and IV.6C).

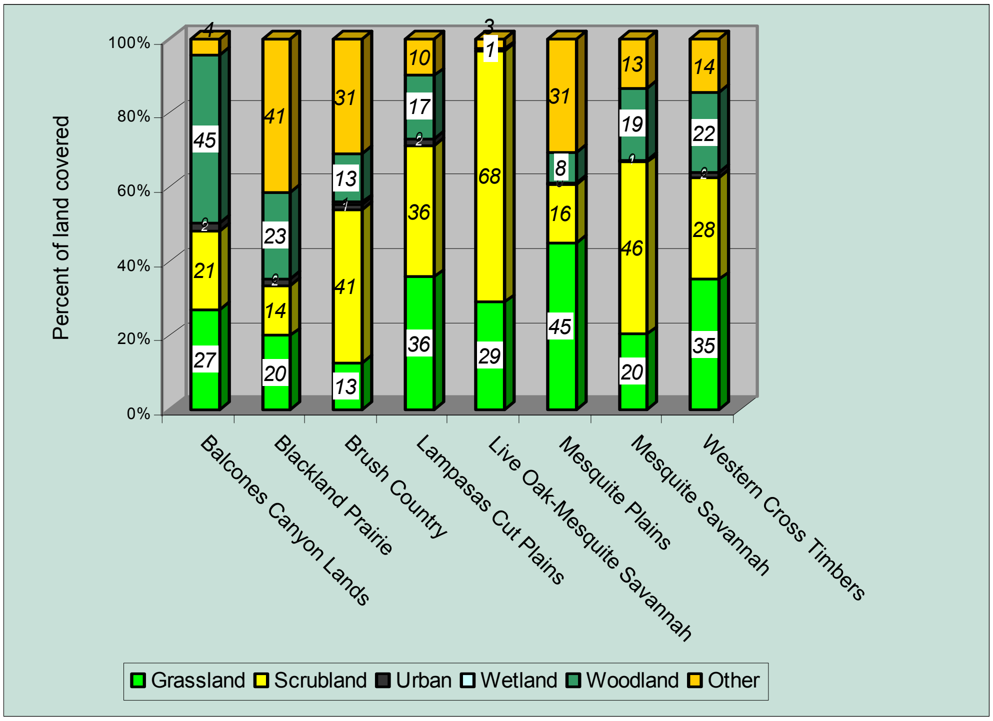

Figure IV.4. *Mean percent of land in each of the 5 land cover classes in each of the 8 ecological sub-regions (Figure III.1) included in the present study.*

Balcones Canyon Lands, Lampasas Cut Plains, Black Land Prairie and Brush Country were the most urbanized sub-regions (PL = 2.2, 1.8, 1.8, and 1.3 respectively). Mean patch size MN (5, 6.5, 4.3, and 3.3 ha, respectively). Proximity index (PROX) was highest in Brush Country (373), followed by Balcones Canyon Lands (281), Lampasas Cut Plains (174), and Black Land Prairie (76), indicating that urban zones were more compact in Brush Country , Balcones Canyon Lands, and Lampasas Cut Plains, Table A-6, Figures IV.4, IV.5D, and IV.6D.

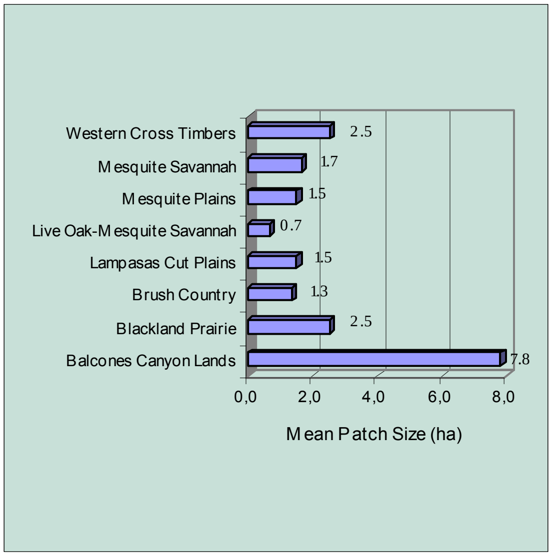

Figure IV.5A. Woodland

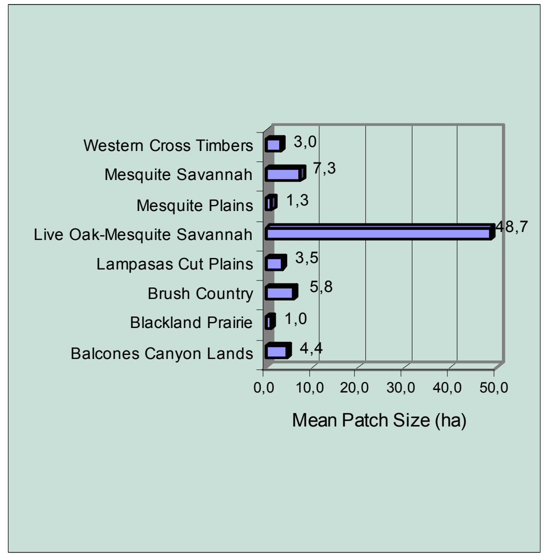

Figure IV.5B. Scrubland.

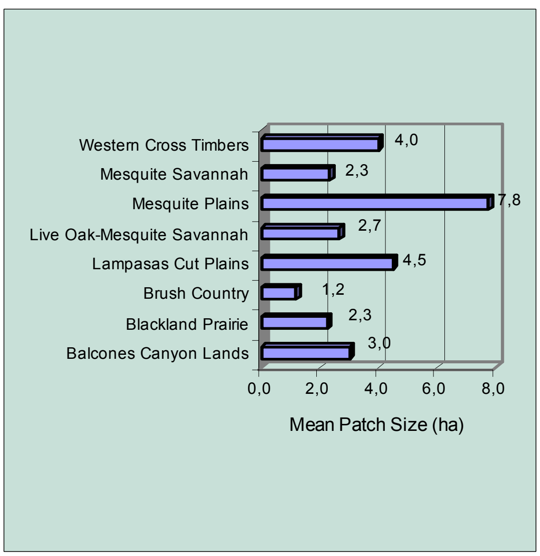

Figure IV.5C Grassland.

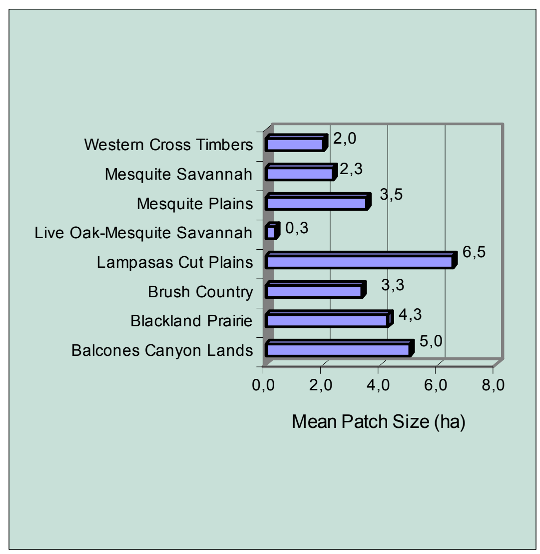

Figure IV.5D. Urban

Figure IV.5. *Mean patch size (MN) of land cover classes (A) woodland, (B) scrubland, (C) grassland and (D) urban, in the ecological subregions*

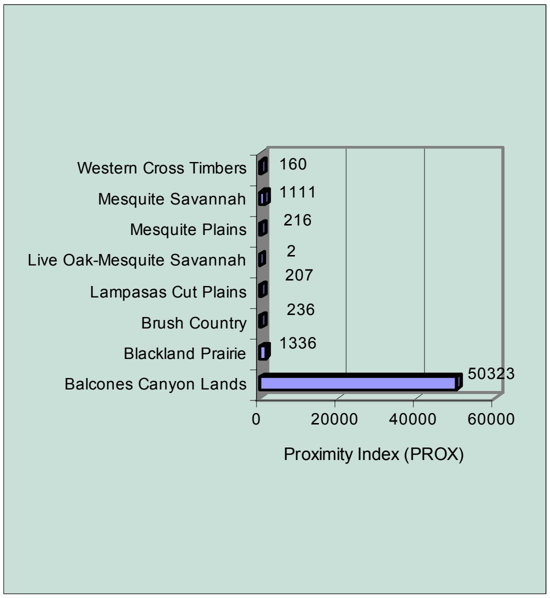

Figure *IV.6*A. Woodland

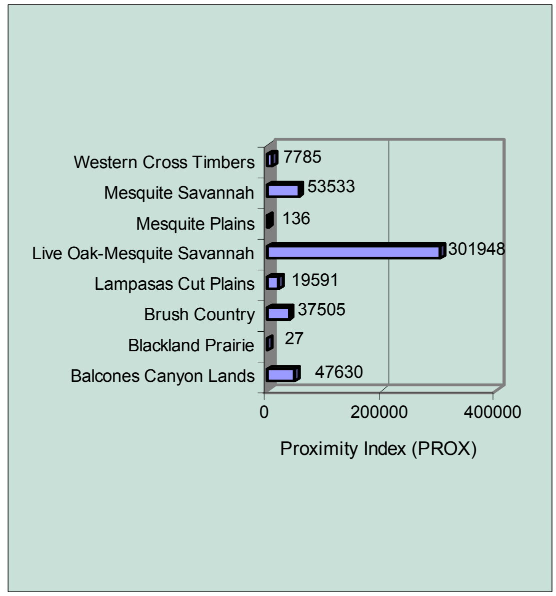

Figure *IV.6*B. Scrubland

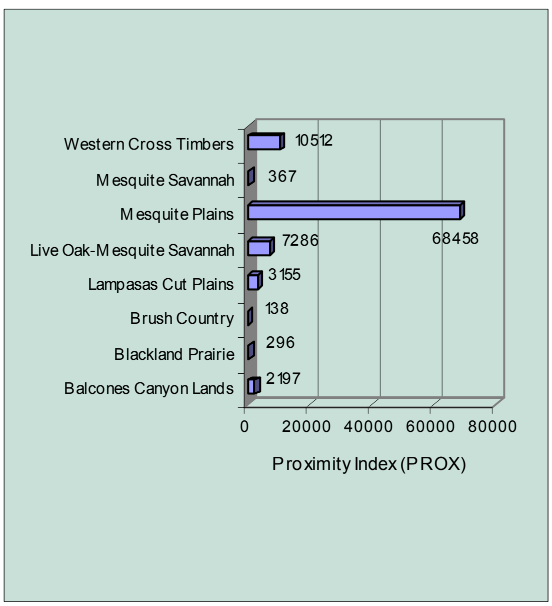

Figure *IV.6*C. Grassland

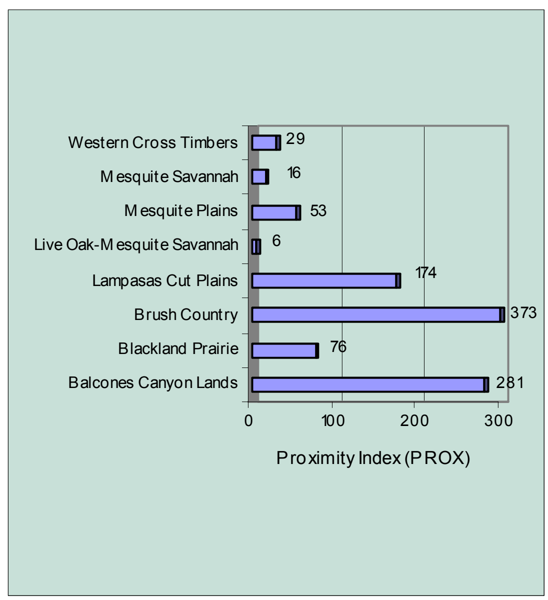

Figure *IV.6*D. Urban

Figure IV.6. *Proximity index of land cover classes (A) woodland, (B) scrubland, (C) grassland and (D) urban, in the ecological subregions.*

Spatial autocorrelation of variables

Approximately half of the indexes showed significant levels of positive spatial autocorrelation, including S2, S3, and S4 ($P = 0.0035$, 0.0013, and 0.0003, respectively), but not S1 ($P = 0.0977$), among ownership property size classes, and SHDI ($P = 0.0072$), but not PR ($P = 0.2406$), among landscape structure indexes at the landscape mosaic level, Table IV.2.

Landscape structure indexes at the land cover class level (PL, PD, MN, and PROX) tended to be more significantly spatially auto-correlated ($P < 0.1$) in scrublands, grasslands, and wetlands than in woodlands and urban areas ($P > 0.1$), although MN was not significantly auto correlated in scrublands ($P = 0.2952$) and PD was significantly auto-correlated in woodlands ($P = 0.0001$).

Spatial correlations between pairs of ownership property sizes and landscape structure indexes

All 4 ownership property size classes (S1 – S4) showed significant positive spatial correlation with both landscape structure indexes at the landscape mosaic level (PR and SHDI) ($P < 0.053$), with one exception: S2 was not significantly spatially correlated with PR ($P = 0.2226$), Table IV.3.
At the land cover class level, the 4 ownership property sizes were more spatially correlated with the 4 landscape structure indexes in scrublands (13 of the 16 pairs showed significant correlations; $P < 0.05$) than in other ecological sub-regions (4/16 in urban areas, 3/16 in woodlands, 2/16 in grasslands, and 0/16 in wetlands); all significant correlations were positive.

Within scrublands, all 4 ownership property sizes were significantly correlated ($P < 0.05$) with PL and PD, all but S2 were significantly correlated with PROX, and S3 and S4 were significantly correlated with MN. Within urban areas, both S2 and S3 were significantly correlated with PL and PD; within woodlands, S1 was significantly correlated with PL, and S3 and S4 were significantly correlated with PD; and within grasslands, both S3 and S4 were significantly correlated with PD.

Table IV.2. *Degree of spatial autocorrelation in each of 4 ownership property sizes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres) and each of 6 indexes of landscape structure (2 at the landscape mosaic level (richness of patches (PR) and Shannon Diversity (SHDI)) and 4 at the land cover class level (percent of land (PL), patch density (PD), mean patch size (MN), proximity (PROX))), as indicated by Mantel's r (Fortin et al., 1993).*

	Indices	Mantel's r	P
Ownership Property Size			
	S1	0.0932	0.0977
	S2	0.2140	0.0035
	S3	0.3418	0.0013
	S4	0.3809	0.0003
Landscape Structure			
Landscape Mosaic Level	PR	0.0498	0.2406
	SHDI	0.2643	0.0072
Land Cover Class Level			
Woodland	PL	0.1009	0.1517
	PD	0.4162	0.0001
	MN	-0.0248	0.5400
	PROX	-0.1049	0.8365
Scrubland	PL	0.2216	0.0071
	PD	0.1303	0.0521
	MN	0.0570	0.2952
	PROX	0.1726	0.0884
Grassland	PL	0.3811	0.0001
	PD	0.1742	0.0416
	MN	0.3734	0.0005
	PROX	0.3140	0.0070
Wetland	PL	0.2164	0.0295
	PD	0.1861	0.0603
	MN	0.2970	0.0063
	PROX	0.1889	0.0605
Urban	PL	-0.0770	0.7763
	PD	0.0696	0.1807
	MN	-0.0643	0.7308
	PROX	-0.0859	0.7581

Table IV.3. *Degree of spatial correlation. Calculation is made between pairs of ownership property sizes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres) and landscape structure ((richness of patches (PR), Shannon Diversity (SHDI), percent of land (PL), patch density (PD), mean patch size (MN), proximity (PROX))) indexes, as indicated by Cross Mantel's r (Fortin et al., 1993).*

		Ownership Property Size (OPS)									
Landscape Structure	Indices	S1		S2		S3		S4		Mean OPS	
		Cross Mantel'r	P	Cross Mantel'r	P	Cross Mantel'r	P	Cross Mantel'r	P	Cross-Mantel'r	P
Landscape Mosaic Level	SHDI	0.2780	0.0014	0.3733	0.0002	0.6573	0.0001	0.6773	0.0001	0,5083	0,0059
	PR	0.1116	0.0523	0.0473	0.2226	0.2720	0.0062	0.2115	0.0163	0,3753	0,0043
Land Cover Class Level											
Woodland	PL	0.1504	0.0372	0.0402	0.3017	0.0932	0.1761	0.1641	0.0681	0,1316	0,11939
	PD	0.1032	0.0934	0.1163	0.0673	0.4467	0.0003	0.4598	0.0001	0,4628	0,0002
	MN	-0.0217	0.4472	-0.0501	0.5944	-0.0313	0.4190	-0.0028	0.3445	-0,0393	0,40886
	PROX	-0.0045	0.4010	-0.0847	0.7697	-0.0680	0.5591	-0.0565	0.5926	-0,0788	0,56014
Scrubland	PL	0.2417	0.0017	0.2532	0.0018	0.4313	0.0005	0.4959	0.0002	0,3683	0,0244
	PD	0.2709	0.0004	0.2377	0.0017	0.1810	0.0385	0.3286	0.0008	0,1549	0,07949
	MN	0.1638	0.1053	0.0082	0.4125	0.3120	0.0279	0.3039	0.0309	0,1272	0,08589
	PROX	0.2695	0.0051	0.1526	0.0690	0.6415	0.0003	0.6150	0.0004	0,5492	0,0305
Grassland	PL	-0.0546	0.7653	-0.0739	0.8519	-0.1292	0.9420	-0.0829	0.8487	-0,0696	0,69273
	PD	0.0845	0.1476	-0.0277	0.6315	0.2731	0.0309	0.2366	0.0325	0,226	0,05889
	MN	-0.0285	0.5778	-0.0833	0.8239	-0.0921	0.7548	-0.0446	0.6045	-0,0388	0,46485
	PROX	0.0015	0.4485	-0.0595	0.7077	-0.0707	0.6207	0.0030	0.3971	0,0199	0,24848
Wetland	PL	-0.1208	0.9687	-0.0459	0.6252	0.0565	0.2312	-0.0829	0.7294	-0,0863	0,59854
	PD	-0.0842	0.8347	0.0418	0.2868	0.1591	0.0937	0.0144	0.2902	-0,0311	0,35716
	MN	-0.1083	0.9387	-0.0759	0.7818	-0.0445	0.5402	-0.0740	0.7174	-0,0147	0,34807
	PROX	-0.1083	0.9425	-0.0950	0.8484	-0.1105	0.8027	-0.1154	0.9213	-0,078	0,59304
Urban	PL	0.2799	0.0015	0.2057	0.0103	0.0087	0.3601	0.1296	0.1080	-0,0309	0,38526
	PD	0.2029	0.0094	0.1573	0.0256	-0.0488	0.6452	0.0678	0.2354	0,0263	0,32517
	MN	0.0362	0.2950	0.0199	0.3717	0.1019	0.1644	0.0358	0.2735	0,006	0,31827
	PROX	0.1367	0.0950	0.1610	0.0571	-0.0815	0.6235	0.0337	0.2387	-0,0762	0,55214

Correlation between values of pairs of ownership property sizes and landscape structure indexes

After appropriate adjustments for spatial autocorrelation, values of all 4 ownership property size classes showed significant correlation ($P < 0.05$) with values of SHDI at the landscape mosaic level, correlations were positive for the 3 smallest size classes (S1 – S3) and negative for the largest size class (S4) and average of ownership property size; only the values of S3 showed a significant (positive) correlation with values of PR, Table IV.4.

At the land cover class level, the values of the 4 ownership property sizes were more correlated with the values of the 4 landscape structure indexes in scrublands (14 of the 16 pairs showed significant correlations) than in other ecological sub-regions (7/16 in urban areas, 3/16 in woodlands, 0/16 in grasslands, and 0/16 in wetlands); significant correlations were both positive and negative. Within scrublands, values of all 4 ownership property sizes were significantly correlated) with values of PL and PROX, values of all but S4 were significantly correlated with values of PD, and values of all but S3 were significantly correlated with values of MN; significant correlations with values of landscape indexes all were negative for values of S1, all were positive for values of S2, and were mixed for values of S3 and S4.

Within urban areas, values of both of the 2 smallest ownership property sizes (S1 and S2) showed significant positive correlations with values of PL, PD, and PROX; the only other significant correlation (between values of S4 and values of PL) was negative. Within woodlands, values of S1 showed significant positive correlations with values of PL, values of S3 showed significant positive correlations with values of PD, and values of S4 showed significant negative correlations with values of PD.

Table IV.4. *Degree of correlation. Calculation is made between values of pairs of ownership property sizes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres) and landscape structure ((richness of patches (PR), Shannon Diversity (SHDI), percent of land (PL), patch density (PD), mean patch size (MN), proximity (PROX))) indexes, as indicated by the Modified t-Test for autocorrelation (CHR) (Clifford et al., 1989; Dutilleul, 1993).*

				Ownership		Property		Size	(OPS)		
Landscape Structure	Indices	S1		S2		S3		S4			Mean OPS
		CHR	P	CHR	P	CHR	P	CHR	P	GRH	P
Landscape Mosaic Level	SHDI	0.5682	0.0403	0.6987	0.0134	0.8149	0.0049	-0.8392	0.0053	-0.6721	0.0241
	PR	0.2971	0.0685	0.2297	0.1525	0.0228	0.8798	-0.1850	0.2000	-0.287	0.0655
Land Cover Class Level											
Woodland	PL	0.4637	0.0543	0.3635	0.1664	0.2745	0.3425	-0.4153	0.1621	-0.4171	0.1129
	PD	0.3805	0.1602	0.3494	0.2556	0.6558	0.0299	-0.5897	0.0382	-0.6212	0.0244
	MN	0.1937	0.3357	0.1397	0.4894	0.0179	0.9327	-0.1175	0.5813	-0.1634	0.4179
	PROX	0.1774	0.3466	0.0837	0.6672	0.2180	0.1787	-0.0155	0.9385	-0.0672	0.7308
Scrubland	PL	-0.5721	0.0227	-0.617	0.0206	-0.5953	0.0444	0.7009	0.0186	-0.337	0.2027
	PD	0.5965	0.0145	0.5997	0.0214	0.4881	0.0992	-0.6464	0.0280	0.0007	0.9975
	MN	-0.3755	0.0718	-0.2665	0.1996	-0.4366	0.0462	0.4479	0.0456	-0.3596	0.153
	PROX	-0.5454	0.0280	-0.4802	0.0711	0.7553	0.0246	0.7283	0.0102	-0.1756	0.3564
Grassland	PL	-0.0383	0.8998	-0.1263	0.7330	0.0324	0.9333	0.0278	0.9462	0.109	0.752
	PD	-0.1023	0.6094	0.0462	0.7959	-0.3057	0.1161	0.1837	0.3516	0.3087	0.1045
	MN	-0.1672	0.5472	-0.1857	0.5682	0.0262	0.9393	0.0976	0.7855	0.0345	0.9105
	PROX	-0.2464	0.3158	-0.1816	0.5124	-0.0682	0.8173	0.1748	0.5654	0.099	0.7142
Wetland	PL	-0.0039	0.9889	0.2235	0.4635	0.3487	0.2681	-0.2532	0.4479	-0.1801	0.5377
	PD	0.1029	0.6914	0.3833	0.1696	0.5046	0.0705	-0.4226	0.1529	-0.2948	0.2668
	MN	-0.0243	0.9171	-0.0129	0.9547	-0.0201	0.9392	0.0290	0.9150	0.0363	0.886
	PROX	0.0872	0.6632	0.0666	0.7225	0.0663	0.7502	-0.0763	0.7160	-0.0679	0.7429
Urban	PL	0.5671	0.0173	0.5379	0.0314	0.3655	0.1987	-0.5540	0.0476	-0.337	0.2027
	PD	0.4994	0.0149	0.3812	0.0356	0.0125	0.9540	-0.2988	0.1610	0.0007	0.9975
	MN	0.2892	0.2468	0.3132	0.1986	0.4785	0.0701	-0.4498	0.0963	-0.3596	0.153
	PROX	0.3502	0.0507	0.3823	0.0382	0.1852	0.3334	-0.3385	0.0711	-0.1756	0.3564

Discussion

Landscape structure is spatially correlated with ownership property sizes in the study area. The correlation was found at land mosaic and class levels. At the class level results showed that there is a threshold of ownership property size of 500 ac, below which habitat fragmentation occurs.

At the land mosaic level, there was positive and significant correlation between SHDI and ownership property sizes smaller than 500 ac, but negative correlation with ownership property sizes larger than 500 ac. I interpret this result to mean that in landscapes with large property sizes, landscape diversity is low, but it increases when land is divided into small properties. In the ecological sub-regions, the Black Land Prairie had the highest SHDI value (1.7) followed by the Western Cross Timbers (1.5). In those ecological subregions, the average of ownership property size (387 and 349 ac, respectively) was below the threshold of 500ac, Table IV.1. That could be because these ecological sub-regions are very close to urban areas where, in addition to natural vegetation patches, there is ornamental vegetation, which can contribute to the landscape diversity.

At the class level, spatial correlation detected with the Cross Mantel is especially clear in scrubland and urban classes of land cover. In scrubland, all indices of land structure showed positive spatial correlation with all ownership property sizes. Correlation identified with Pearson Correlation and modified *t*-test, is significantly negative between percent of land (PL) and MN with ownership property sizes smaller than 500 ac. PROX became positive with ownership property sizes larger than 100 ac. It means that patches of habitat begin to decrease when land is dominated by ownership property sizes smaller than 500 ac, but the number of scrubland patches with distance of 100 m. in between them decreases if land becomes dominated by ownership property sizes smaller than 100 ac. In urban PL, and PD, showed positive spatial correlation with ownership property sizes smaller than 100 ac., which is an inverse to the correlations with structure indexes of classes of natural land cover (scrubland).

At both land mosaic and class levels, the tendency of the correlation changed at 500 ac ownership property sizes; so, it seems as if it corresponds to a threshold in size of ownership below of which the landscape fragmentation increases. It represents an increase in diversity at land mosaic and habitat fragmentation at class level, which is indicated by decrease of mean patch size, and greater proximity among patches.

Ownership property size of 500 ac is a threshold too, for consolidation of urban or suburban landscape. It is indicated by the positive and significant correlation among all the urban structure indexes and percentage of land occupied by ownership properties smaller than 100 acres, and negative over 500 ac, Table IV.4. The inverse tendency of the correlations with respect to those with class structure indexes of natural habitats as scrubland, can indicate that natural habitats have been fragmented by the increase in urbanization.

In the study area, there are seven counties with an average of ownership property size below of the threshold of 500 ac, and 6 very close to it. Counties with ownership property sizes below the threshold are: Karnes (364 ac) and Wilson (281 ac) in the Brush Country sub-ecoregion; Caldwell (276 ac) and Falls (366 ac) in the Lampasas Cut Plains; and Hood (343 ac.) in the Mesquite Plains, Table A-7. All of those counties have above 75% of the area covered by ownership property sizes smaller than 500 ac, Figure IV.4.

Counties with an average of ownership property size very close to that of the threshold of 500 ac, are: Kendall (528 ac), Hays (658 ac) in Balcones Canyon Land; Coryell (609 ac.) and Lampasas (628 ac), in Lampasas Cut Plains; and Palo Pinto (706 ac) in Mesquite Plains. All of those counties have around 70% of the area covered by ownership property sizes smaller than 500 ac Figure IV.4.

Identification of the 500-ac threshold, is important for conservation planning because decrease in habitat patch size and proximity among patches, are factors that are directly related with fauna meta-populations management, because isolated patches act as habitat islands which colonization or repopulation depends on the patch area and proximity among patches.

Conclusions

Landscape structure is spatially correlated with ownership property sizes in the study area. The correlation existed both at the land mosaic level and at the class level. Significant spatial correlations detected by Cross Mantel and significant correlations detected by Pearson correlation and modified *t*-test, identified a threshold of ownership property size at 500 ac, below of which, land diversity increases at the land mosaic level, but habitat fragmentation increases at the class level. The latter effect is represented by a decrease in mean patch size, and an increase in the distance between equivalent patches.

In general the threshold of ownership property size, 500 ac, is important for conservation planning because below that threshold there begins to be decrease in habitat patch size and increase in distance between equivalent patches of habitat. Those are factors directly important to faunal meta-population management, because isolated patches act as habitat islands, within a sea of less suitable habitat.

In the study area there are 7 counties with an average ownership property size below of the threshold of 500 ac, and 6 counties with an average ownership property size close to the threshold of 500 ac. Counties with ownership property sizes below the threshold are 2, Karnes (364 ac) and Wilson (281 ac) in Brush Country sub-ecoregion; Caldwell (276 ac) and Falls (366 ac) in Lampasas Cut Plains; and Hood (343 ac.) in Mesquite Plains.

Counties with average of ownership property size close to the threshold of 500 ac are: Kendall (528 ac), Hays (658 ac) in Balcones Canyon Land; Cory ell (609 ac.), and Lampasas (628 ac), in Lampasas Cut Plains; Palo Pinto (706 ac) in Mesquite Plains.

CHAPTER V

SPATIAL RELATIONSHIPS BETWEEN URBAN INFLUENCE AND OWNERSHIP PROPERTY SIZE, WITHIN THE EDWARDS PLATEAU ECOREGION, TEXAS: IMPLICATIONS FOR CONSERVATION PLANNING

Introduction

Human-induced landscape transformations have important implications for the maintenance of biodiversity. Ecological processes are related not only to land use, but also to landscape structure, that is, to the spatial arrangement of land elements (Zonneveld and Forman 1989; Baudry, 1993). Among the more pervasive landscape changes whose impact on biodiversity is potentially great, yet remains poorly understood, is that associated with changes in ownership property sizes along urban – rural gradients. Urban sprawl generates economic pressure extending well past city limits into the rural landscape (Costanza et al., 1997), which leads to a reduction in ownership property sizes (Adger and Luttrel 2000; Antrop 2000; Swenson and Franklin 2000; Luck and Wu 2002).

The urban "fringe" is that part of metropolitan counties that is not settled densely enough to be called "urban." Low-density development (2 or fewer houses per acre) of new houses, roads, and commercial buildings causes urban areas to grow farther out into the countryside, and increases the density of settlement in formerly rural areas (USDA 1992), especially those that are scenic or offer recreational opportunity. In the urban fringe, rural landowners expect increased land prices, because taxes in such areas tend to rise as urbanization occurs (Alig et al. 2004).

In Texas, many rural areas are experiencing greatly increased residential development; the southern and western portions of the state (the Trans Pecos, Edwards Plateau, South Texas Brush Country, and Coastal Sand Plains ecoregions) have been loosing annually more than 235,000 acres that were in large ownerships (>2000 acres), thus dramatically shifting the size-class distribution of ownership property sizes within these regions (Wilkins et al. 2003). This is particularly important because the Edwards plateau ecoregion, has been declared a biodiversity hot spot (Myers et al. 2000).

In this study, I investigated linkages between urban influence and ownership property size within the Edwards Plateau Ecoregion of Texas, with particular emphasis on the spatial correlation of urban influence index to ownership property sizes and its implications in conservation planning. More specifically, I tested for statistically significant spatial correlations between urban influence index and each of 4 ownership property sizes.

Study area

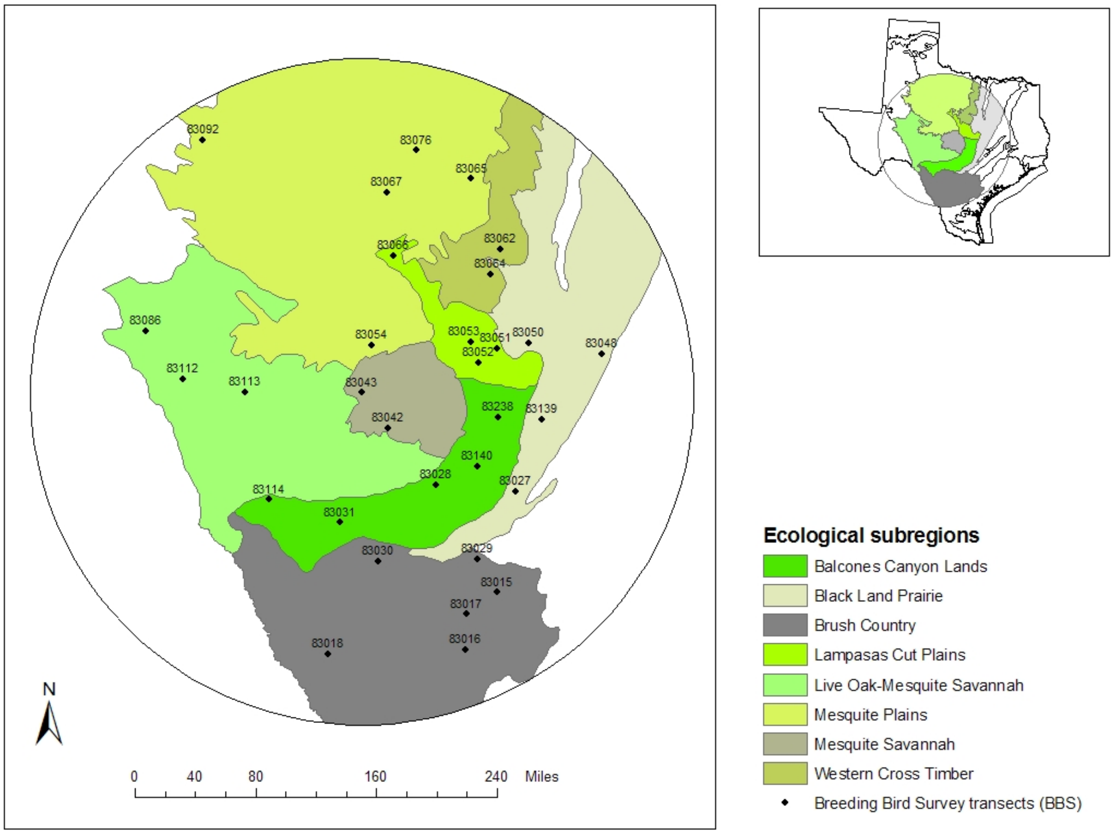

Figure V.1. *Geographic location of the study area. The study area included the 8 ecological sub-regions referred to in this dissertation. Locations of the 31 North American Breeding Bird Survey (BBS) transects included in the present study also are indicated; numbers correspond to the BBS code, of which the first two digits (83) signify Texas, and the final three digits (015-238) correspond to the transect number.*

The study area was bounded by a circle of radius 300 km centered at the geographical center point of the Edwards Plateau Ecoregion, which includes all of the Edwards Plateau and portions of 5 other ecoregions; South Texas Brush, Blackland Prairie, Llano Uplift, Rolling Plains, and Oak Woods.

For purposes of the present study, I used the 8 Texas ecological subregions (Gould 1975. adapted by Wu et al. 2002), included in these ecoregions: Balcones Canyon Lands, Black Land Prairies, Brush Country, Lampasas Cut Plains, Live Oak-Mesquite Savannah, Mesquite Savannah, Mesquite Plains, and Western Cross Timbers (Figure V.1).

The climate ranges from subtropical steppe to subtropical sub-humid, with mean annual precipitation ranging from 375 mm in the west to 750 mm in the east, about three-fourths of which falls during the growing season (April through mid-November). The area is predominantly shrub land grazed by cattle, sheep, and goats, but local tracts are cultivated for domestic pasture and hay; cotton and grain sorghum are grown locally on irrigated land and there are some pecan orchards on flood plains. Landowners commonly lease their land for hunting deer, quail, mourning dove, wild turkey, and/or javelina. Many rural areas are experiencing greatly increased residential development, especially in the eastern portion of the region, due in large part to the influence of large cities such as San Antonio and Austin (Wilkins et al. 2003).

Methods

I first describe, for each variable (urban influence and ownership property sizes), the databases, the calculation of the appropriate indexes, and the creation of the appropriate shape files. Second, I describe the test for spatial autocorrelation for each variable (urban influence and ownership property sizes). Third, I describe the tests for spatial correlations between each pair of variables (Cross Mantel Test). Fourth, I describe the test for correlation between each pair of variables (the Modified t-Test for autocorrelation). Finally, I describe the spatial structure of the urban influence index, and the percent of land in four ownership property sizes, in the ecological subregions of the Edwards Plateau.

Urban influence

I obtained population data for cities that had >30 inhabitants in 1992 within a 300 km radius circle around the geographical center point of the Edwards Plateau Ecoregion. I used the information from the National Atlas of the United States of America (1997).Then, I calculated an index of urban influence associated with each of the midpoints of the 31 BBS transects:

$$UI_j = \sum \frac{P_i}{D_i^2} \qquad \text{(V.1)}$$

where UI_j represents the index of urban influence at the midpoint of the jth BBS transect, P_i represents the number of inhabitants of the ith city, and D_i represents the distance (km) between the ith city and the midpoint of the jth BBS transect; UI has been used to represent the influence of cities on the market value of rural lands (Jin Shi et al., 1997). Finally, I calculated the average value of the urban influence index for the ecological sub-regions.

Ownership property sizes

I obtained data on ownership property sizes in 1992 for each of the 27 Texas counties that contained one or more of the 31 North American Breeding Bird Survey (BBS) transects described in Chapter II. These data included average size of rural (farm and ranch) property sizes in acres (USDA, 1992) and the proportions of rural acreage in each of 4 size classes: <50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres (unpublished data, Land Information Systems Laboratory, Texas A&M University; data summary available in Wilkins et al. (2003). I also calculated the mean and standard deviation of the proportions of acreage in each of the 4 size classes in each of the 8 ecological sub-regions.

Spatial autocorrelation of variables

I tested for spatial autocorrelation in each of the urban influence index and ownership property sizes using a Mantel Tests (Fortin and Gurevitch 1993). The Mantel Test (r) is a regression in which the variables themselves are distance, or dissimilarity (ecological distances), matrixes summarizing pair-wise similarities among sample locations:

$$r = \frac{\sum\sum stdA_{ij} * stdB_{ij}}{n-1} \text{ (sum from i to n and sum from j to n, for i} \neq \text{j)} \qquad \text{(V.2)}$$

where n is the number of sample locations, i and j identify the matrix element, B_{ij} is the Euclidian distance matrix of location points, and A_{ij} is the dissimilarity matrix of the variable of interest, the present case, the urban influence index and S1, S2, S3, S4, For Mantel Test calculations, I used the PASSAGE program (Rosenberg, M. 2005).

Spatial correlations between urban influence index and ownership property sizes.

To determine if the index of urban influence and ownership property sizes are spatially correlated, I conducted a Cross Mantel Test (r) (Fortin and Gurevitch 1993), between each pair of urban influence index and ownership property sizes. This set of analyses tests for a spatial relationship *per se* between differences of pairs of values of both variables (avian diversity and landscape structure), but does not indicate the degree of correlation between the values of the two variables. For example, if the differences in the values of variable A and the differences in the values of variable C increase or decrease with increasing distance (that is, with increasing geographical distance between the points where the values of the variables were measured), the variables A and C are positively spatially correlated. If the differences in the values of variable A increase with distance and the values of variable C decrease with distance, or vice versa, then variables A and C are negatively spatially correlated.

$$r = \frac{\sum\sum stdA_{ij} * stdC_{ij}}{n-1}$$ (sum from i to n and sum from j to n, for i ≠ j) (V.3)

where n is the number of sample locations, i and j identify the matrix element, A_{ij} is the dissimilarity matrix of one of the variables of interest, that is the urban influence index, , and C_{ij} is the dissimilarity matrix of the other variable of interest, that is an ownership property size (S1, S2, S3, S4),

Correlation between values of urban influence index and ownership property sizes

I conducted a Pearson's pair-wise correlation (ρ) between each pair of urban influence indexes and ownership property sizes. This set of analyses identifies the degree of correlation between the values of the two variables,

$$\rho = \frac{\sum_{i=1}(u_i - m_u)(v_i - m_v)}{S_u S_v} \quad \text{(V.4)}$$

where u and v are two variables (u is the urban influence index and v is one of the 4 ownership property sizes, m_u and m_v are their respective means, and S_u and S_v are their respective standard deviations. I then used the Modified *t*-test for autocorrelation (CHR), which corrects the degrees of freedom based on the amount of autocorrelation in the data, to assess the correlation between each pair of spatially correlated variables (Clifford et al. 1989; Dutilleul 1993). The procedure calculates the amount of spatial autocorrelation of variables to determine how different the effective sample size (n') is from the number of observations.

$$n'(R) = \frac{n^2}{\sum\sum cor(u_i u_j)} \qquad \text{(V.5)}$$

where R is the autocorrelation matrix, n is the number of observations, and u_i, and u_j are the observations of the two variables. The corrected degrees of freedom (n’-2) were then used to test the significance of the correlation; I used the PASSAGE program (Rosenberg 2005) to perform these calculations

Results

Urban influence in the study area

Urban influence index based on values of UI_j on the 31 transects, showed a gradient east to west in the study area, with values ranging from 0.00004 (BBS transect 83092) to 0.0012 (BBS 83238), Table A-7. High values in the east of the ecoregion, are contiguous to the Austin-San Marcos-San Antonio metropolitan corridor, Figure V.2.

At the level of ecological subregion, the average of the urban influence index was, from highest to lowest values, Balcones Canyon Lands and Black Land Prairie (0,0005); Lampasas Cut plains, Brush Country, and Western Cross Timber (0.0003); and Live Oak-Mesquite Savannah, Mesquite Plains and Mesquite Savannah (0.0001), Table V.1.

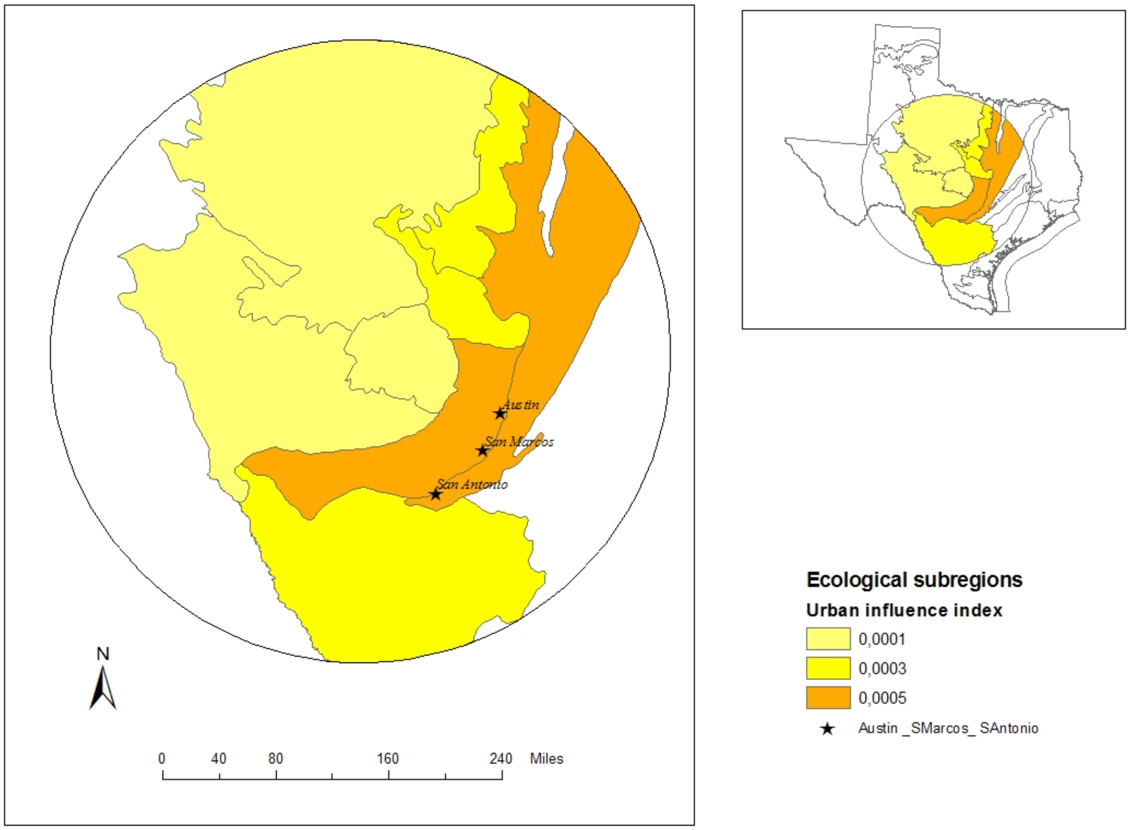

Figure V.2. *Urban influence in the study area by ecological subregion. Austin-San Marcos-San Antonio Metropolitan corridor location.*

Table V.1. *Mean, maximum, and minimum of 4 ownership property sizes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) in acres, the range of sizes (maximum - minimum), and urban influence index. Calculation is made in each of the 8 ecological subregions included in this study.*

Ecological subregion	Data	Mean OPS	S1	S2	S3	S4	Urban influence index
Balcones Canyon Lands	Mean	1441	20	10	34	36	0.0005
	Max	4232	28	17	41	69	0.0011
	Min	298	8	2	21	17	0.0001
Blackland Prairie	Mean	387	20	16	45	20	0.0005
	Max	609	28	19	52	28	0.0008
	Min	276	12	9	38	14	0.0003
Brush Country	Mean	965	17	12	43	28	0.0003
	Max	3288	25	18	55	55	0.0008
	Min	281	10	6	29	11	0.0001
Lampasas Cut Plains	Mean	625	17	10	46	29	0.0003
	Max	636	22	11	52	31	0.0003
	Min	609	12	9	42	25	0.0002
Live Oak-Mesquite Savannah	Mean	7305	7	4	14	75	0.0001
	Max	12746	13	6	19	90	0.0001
	Min	2964	3	2	4	66	0.0001
Mesquite Plains	Mean	1530	13	9	39	40	0.0001
	Max	2391	22	13	46	56	0.0003
	Min	706	8	6	30	26	0.0000
Mesquite Savannah	Mean	1089	9	4	37	50	0.0001
	Max	1278	9	4	41	52	0.0002
	Min	995	9	4	35	46	0.0001
Western Cross Timbers	Mean	349	25	17	42	16	0.0003
	Max	355	29	19	47	18	0.0003
	Min	343	21	14	37	14	0.0003

Ownership property sizes

Mean ownership property sizes for counties ranged from 276 acres (Caldwell County) to 12,746 acres (Crocket County) (Table A-7), increasing along an east-west gradient (Figure V.3). Largest average of ownership property sizes by ecological subregion, occurred in Live Oak-Mesquite Savannah, followed by Mesquite Plains and Balcones Canyon Lands, Mesquite Savannah, Brush Country, Lampasas Cut Plains, and Western Cross Timber , with maximum and minimum sizes, and ranges between maximum and minimum sizes, all decreasing in generally the same order, with a few exceptions (Table V.2).

Although a relatively high percentage of the rural acreage was in the largest (S4, >500 acres; from 16 to 75%) and next-to-largest (S3, 100-500 acres; from 14 to 45%) size classes in all

8 ecological subregions, 7 of the 8 subregions also had roughly one-fifth of the rural acreage in the smallest (S1, <50 acres; from 7 to 25%) size class, the exception being Live Oak-Mesquite Savannah (only 7% in S5) (Figure V.4).

Table V.2. *Mean, maximum, and minimum ownership property sizes (OPS) in acres. The range of sizes (maximum - minimum), for counties in each of the 8 ecological subregions included in this study.*

Ecological subregion	Data	Mean OPS	S1	S2	S3	S4
Balcones Canyon Lands	Mean	1441	20	10	34	36
	Max	4232	28	17	41	69
	Min	298	8	2	21	17
Blackland Prairie	Mean	387	20	16	45	20
	Max	609	28	19	52	28
	Min	276	12	9	38	14
Brush Country	Mean	965	17	12	43	28
	Max	3288	25	18	55	55
	Min	281	10	6	29	11
Lampasas Cut Plains	Mean	625	17	10	46	29
	Max	636	22	11	52	31
	Min	609	12	9	42	25
Live Oak-Mesquite Savannah	Mean	7305	7	4	14	75
	Max	12746	13	6	19	90
	Min	2964	3	2	4	66
Mesquite Plains	Mean	1530	13	9	39	40
	Max	2391	22	13	46	56
	Min	706	8	6	30	26
Mesquite Savannah	Mean	1089	9	4	37	50
	Max	1278	9	4	41	52
	Min	995	9	4	35	46
Western Cross Timbers	Mean	349	25	17	42	16
	Max	355	29	19	47	18
	Min	343	21	14	37	14

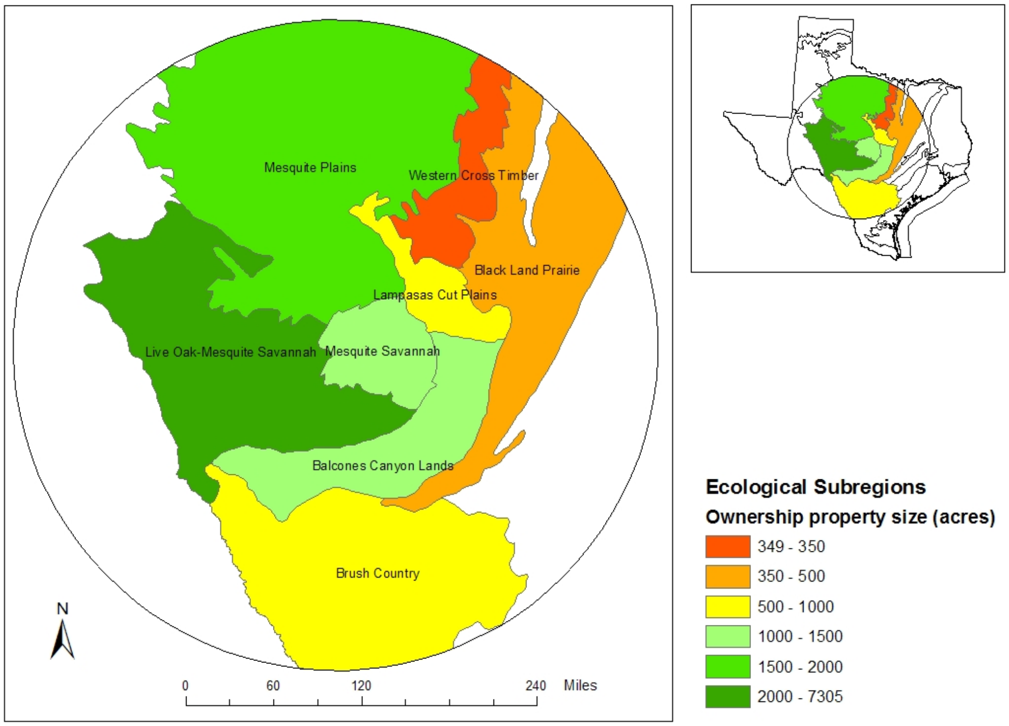

Figure V.3. *Ownership property size in the ecological subregions.*

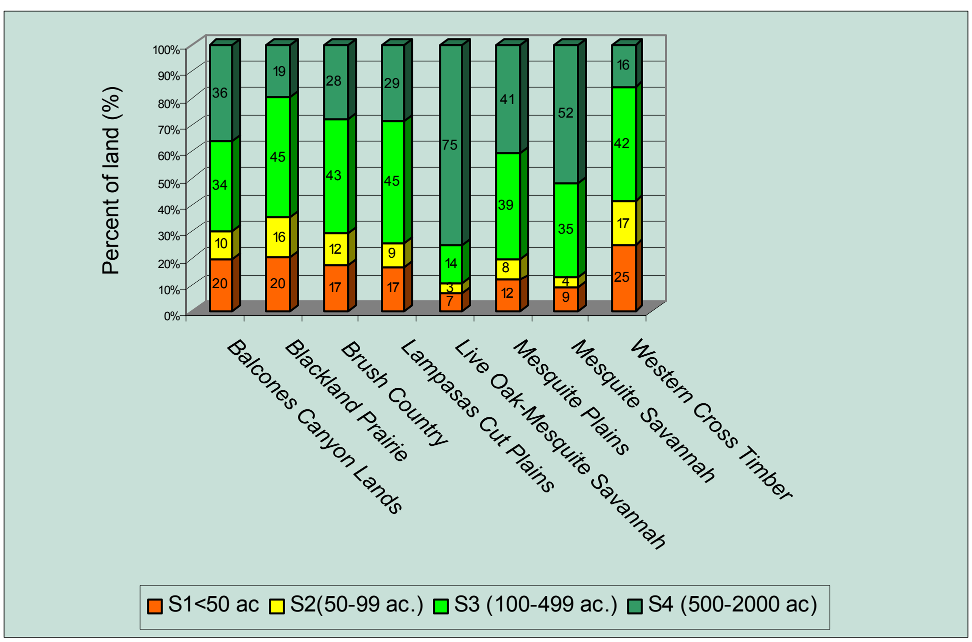

Figure V.4. *Mean percent of rural acreage in each of 4 size classes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres) for counties within each of the 8 ecological subregions included in this study.*

Spatial autocorrelation of variables

Urban influence index and ownership property sizes, showed a significant positive autocorrelation. It means that both variables have a spatial pattern in the study area. Ownership property sizes smaller than 50 acres had a small value of the autocorrelation, but other classes showed high autocorrelation. Urban influence index showed a high autocorrelation value, Table V.3.

Table V.3. *Degree of spatial* autocorrelation of urban influence index and *4 ownership property sizes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4), as indicated by Mantel's r (Fortin and Gurevitch 1993)*

Ownership property size	Mantel's r	P
S1	0.0932	0.0977
S2	0.2140	0.0035
S3	0.3418	0.0013
S4	0.3809	0.0003
Urban Influence Index	0.3296	0.0081

Spatial correlations between urban influence index and ownership property sizes.

Cross Mantel Test results showed a highly significant positive spatial correlation, between urban influence index and ownership property sizes, Table V.4.

Table V.4. Degree of spatial correlation. Calculation is made between urban influence index and 4 ownership property sizes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres), as indicated by Cross Mantel's r (Fortin and Gurevitch 1993).

	Ownership property sizes							
	S1		S2		S3		S4	
	Cross_Mantel'r	P	Cross_Mantel'r	P	Cross_Mantel'r	P	Cross_Mantel'r	P
Urban Influence Index	0.4484	0.0001	0.4238	0.0001	0.0072	0.36836	0.2675	0.0285

Correlation between values of urban influence index and ownership property sizes

Modified t-test confirmed the correlation between both variables, but showed that the positive correlation with percent of land occupied with ownership property sizes smaller than 499 acres become negative with percent of land occupied with ownership property sizes taller than 500 acres. It is the same ownership property size at which in Chapter III, was found the threshold of habitat fragmentation (Table V.5).

Table V.5. *Degree of correlation. Calculation between urban influence index and 4 ownership property sizes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) acres). as indicated by Modified t-Test for autocorrelation (CHR) (Clifford et al. 1989; Dutilleul 1993).*

	Ownership property sizes							
	S1		S2		S3		S4	
	CHR	P	CHR	P	CHR	P	CHR	P
Urban Influence Index	0.7205	0.0038	0.7076	0.0092	0.3827	0.2516	-0.665	0.0356

Discussion

In the study area, urban influence and ownership property size were spatially correlated. Positive correlation between urban influence and ownership property sizes smaller than 500 ac is a highly important result since the conservation diversity planning view, because ownership property sizes smaller than 500 ac (fragmentation threshold) are correlated with habitat fragmentation as was demonstrated in Chapter IV. So urban growth of the Austin-San Marcos-San Antonio metropolitan belt in the northwest way over the Edwards Plateau ecoregion, should have had an effect on the landscape structure, on the habitats and on the bird diversity.

Ownership property sizes smaller than 500 ac were geographically located in Western Cross Timber, Black Land Prairie, Lampasas Cut Plains and Brush Country (Figure V.3). In those ecological sub regions, ownership property sizes smaller than 100 acres, represented at least 30% and ownership property sizes between 100 to 499 acres represented 42 to 46% (Figure V.4).

The fact that Lampasas Cut Plains showed similar proportions of smaller than 500 ac ownership property sizes with respect to ownership property sizes > 2000 ac, as Brush Country and Black Land Prairies, can indicate that the eastern portion of the Edwards Plateau ecoregion had been affected by a faster parcelization process (Alig et al. 2004), because the Austin-San Marcos metropolitan belt, which has been especially growing in the north-west direction far away from the IH 35 (Cowley and Naik 2001), it seems the same parcelization process identified for main US metropolitan areas in which population growth and household formation, combined with growth in income and wealth, spurs new housing development and consumption of land for housing is occurring in these counties (Alig et al. 2004; USDA 1992).

Usually people in those areas like to live 30 miles away from cities with over 50,000 inhabitants (USDA Census Bureau 1992), especially selecting those rural areas attractive for recreational activities. Far in advance of the urbanization process, landowners have expectations about the increase of rural land prices, because taxes increase for lands around the areas undergoing urbanization increase (Alig et al. 2004).

Perhaps more critically, Balcones Canyon Lands and Mesquite Savannah subregions, where the ownership property mean size already was close to the threshold for habitat fragmentation (500 ac), and the proportion of ownership property sizes smaller than 500 ac exceeded 50%, can be expected to undergo even further fragmentation, because of the urban growth along the west side of IH 35. This is important to take in account, because Balcones Canyon Lands is the ecological subregion of the Edwards Plateau, which had the highest richness and α -diversity of birds (Chapter III Conclusions), and the Edwards Plateau ecoregion, had been declared a biodiversity hot spot (Myers et al. 2000).

The growth in proportion of small ownership property sizes versus big ownership property sizes, can be a good indicator of the parcelization process, because urban influence is positively correlated with small ownership property sizes, and simultaneously, they are positively correlated with habitat fragmentation (Chapter IV, Conclusions). Ownership property size at 500 acres, can be used as indicator of the “parcelization wave”, extending far away from the Austin-San Marcos-San Antonio metropolitan corridor, over the Edwards Plateau. It is important to monitor the proportion among small and big ownership properties in Lampasas Cut Plains and Balcones Canyon Lands, east ecological subregions of the Edwards Plateau,

Conclusions

Urban influence and ownership property size are spatially correlated in the Edwards Plateau Ecoregion. The significant positive correlation between urban influence and ownership property sizes smaller than 500 acres, became negative with ownership property sizes bigger than 500 acres. Ownership property sizes smaller than 500 acres were geographically concentrated along the eastern border of the Edwards Plateau.

Highest percent of ownership property sizes smaller than 500 acres in east portion of the study area, Western Cross Timbers, Black Land Prairie, Lampasas cut plains and Brush Country), could reflect a faster parcelization process in this subregion, because the urban growth in the Austin-San Marcos metropolitan area, over scenic places in this sub-ecoregion.

Because Balcones Canyon Lands show the same tendency of parcelization as Lampasas Cut Plains, it is important to monitor the parcelization process, because the Edwards plateau is the biodiversity heart of Texas, and this process contributes to habitat fragmentation, which directly impacts wildlife communities.

CHAPTER VI

URBAN INFLUENCE ON α -DIVERSITY and β - DIVERSITY OF AVIFAUNA, SPATIAL RELATIONSHIPS IN THE EDWARDS PLATEAU OF TEXAS

Introduction

In exploratory studies at the level of the 76 ecoregions of North America, the Edwards Plateau in Texas has been classified as a high conservation-priority ecoregion, due to spatial coincidence of high biological diversity and agriculture (Ricketts and Imhoff 2003). In this classification other Texas ecoregions, such as Black Land Prairies, are recognized for conservation priority because there is spatial coincidence of high biodiversity and several other factors, in this case agriculture and urbanization.

Although two of the big metropolitan areas of Texas, Dallas-Fort Worth and Austin-San Marcos, are located inside the geographical limits of Black Land Prairies ecoregion, the main area of concern for the Austin-San Marcos area is in the Hill Country, that is, on the Edwards Plateau. It is highly probable that the urban expansion of this metropolitan area will have effects upon the biological diversity of this ecoregion.

Some studies using birds as indicators have shown that urbanization affects landscape heterogeneity, and consequently the distribution and abundance of resources the birds use for their sustenance (Blair 2004; Donovan and Flather 2002; Luck and Wu 2002). Therefore, in the first stage, at a moderate level of urban development, avian diversity reaches its peak in the area, because of the establishment of ornamental vegetation zones; the increase of between-habitat borders; and, the higher water availability, all of which increase landscape heterogeneity. However, at an extreme level of urban development, avian diversity decreases as landscape heterogeneity and resource availability decrease along with the replacement of natural features by concrete and urban structures (McKinney 2002.).

Surprisingly, studies concerning this topic for the Edwards Plateau are scant. In the popular literature are several reports of urbanization’s negative effect on biological diversity of the Edwards Plateau (Hillis 2000), but this is not the case for the scientific literature. Therefore, it is very important to evaluate this relationship from a scientific point of view, so that the future environmental management plans, focusing on biological diversity conservation, take into account not only agriculture but also urban growth in the surrounding areas of Edwards Plateau.

I investigated linkages between urban influence and bird diversity within the Edwards Plateau Ecoregion of Texas, with particular emphasis on the spatial correlation of urban influence index to bird diversity indexes. More specifically, I tested for statistically significant spatial correlations between (1) an index of urban influence and each of 3 indexes of avian α -diversity, and 1 index of avian β -diversity.

Study area

The study area was bounded by a circle of radius 300 km centered at the geographical center point of the Edwards Plateau Ecoregion, which includes all of the Edwards Plateau and portions of 5 other ecoregions; South Texas Brush, Blackland Prairie, Llano Uplift, Rolling Plains, and Oak Woods.

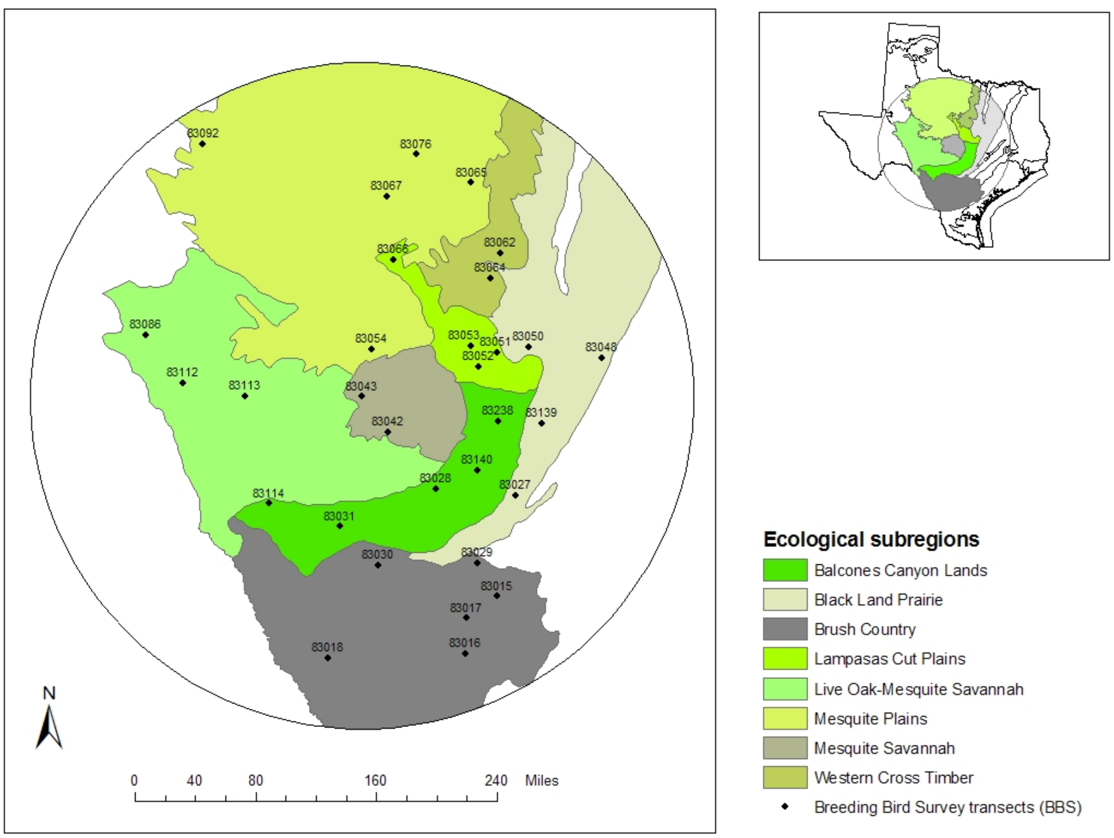

Figure VI. 1. Geographic location of the study area. The study area included the 8 ecological sub-regions referred to in this dissertation. Locations of the 31 North American Breeding Bird Survey (BBS) transects included in the present study also are indicated; numbers correspond to the BBS code, of which the first two digits (83) signify Texas, and the final three digits (015-238) correspond to the transect number.

For purposes of the present study, I used the 8 Texas ecological subregions (Gould 1975. adapted by Wu et al. 2002), included in these ecoregions: Balcones Canyon Lands, Black Land Prairies, Brush Country, Lampasas Cut Plains, Live Oak-Mesquite Savannah, Mesquite Savannah, Mesquite Plains, and Western Cross Timbers (Fig VI.1).

The climate ranges from subtropical steppe to subtropical sub-humid, with mean annual precipitation ranging from 375 mm in the west to 750 mm in the east, about three-fourths of which falls during the growing season (April through mid-November). The area is predominantly shrub land grazed by cattle, sheep, and goats, but local tracts are cultivated for domestic pasture and hay; cotton and grain sorghum are grown locally on irrigated land and there are some pecan orchards on flood plains. Landowners commonly lease their land for hunting deer, quail, mourning dove, wild turkey, and/or javelina. Many rural areas are experiencing greatly increased residential development, especially in the eastern portion of the region, due in large part to the influence of large cities such as San Antonio and Austin (Wilkins et al. 2003).

Methods

I first describe the databases and the calculation of the indexes used to represent avian diversity and urban influence. Second, I describe the test for spatial autocorrelation for each variable (index). I then describe the tests for spatial correlations between each pair of variables. Finally, I describe the test for correlation between the values of each pair of variables, corrected for effects of spatial correlation.

Avian diversity indexes

I assembled avian data collected from 1990 to 1994 from 31 North American Breeding Bird Survey (BBS) transects (ranging in length from 35.7 to 39.3 km), including 12 transects located within the three ecological subregions of the Edwards Plateau Ecoregion (Live Oak-Mesquite Savannah, Balcones Canyon Lands, and Lampasas Cut Plains) and the 19 closest transects located in adjacent ecological subregions: 4 in Blackland Prairie, 2 in Mesquite Savannah, 2 in Western Cross Timber, 5 in the Mesquite Plains, and 6 in the Brush Country (Fig. VI.1); no data were available for ecological subregions of the Trans Pecos Ecoregion, and data were not available for all years on some 31 transects, but each transect had at least 3 consecutive years of data. I chose transects in adjacent ecoregions to represent an “ecological border” (as defined by Cadenasso et al. 2003). I included only data from 1990 to 1994 for two reasons: land cover data for the Edwards Plateau were available only for 1992 (see Section 3.2), and the landscape of the Edwards Plateau was changing rapidly during this period (Wilkins et al. 2003; see Introduction).

The BBS, which began in 1966, consists of a set of roadside surveys (over 3,500 transects have been established) conducted each June by experienced birders to provide an index of population change for songbirds (Sauer et al. 2003). Data include the number of species of birds and the number of individuals of each species observed on each transect. Each species also is characterized in terms of habitat preferences (e.g., grassland) and migration status (e.g., neotropical migrant).

I then calculated, for each transect, abundance (total number of individuals of all species per 10 km of transect), two indexes of α-diversity (species richness (number of different species per 10 km of transect) and Shannon's diversity (H')), and an index of β-diversity (1 - S) (Ludwig and Reynodsl., 1988), each averaged over the period from 1990 to 1994.

$$H' = -\sum \left(\frac{n_i}{N} \right) * \ln \left(\frac{n_i}{N} \right) \qquad \text{(VI.1)}$$

where n_i represents the number of individuals of the ith species per 10 km of the transect and N represents the total number of individuals of all species per 10 km of the transect.

$$\beta\text{-diversity} = 1 - S \qquad \text{(VI.2)}$$

$$\text{where } S = (2 * pN) / (aN + bN) \qquad \text{(VI.3)}$$

and aN and bN represent the total number of individuals of all species per 10 km of transect in the first and second transects, respectively, and pN represents the sum of the lower of the 2 abundances for each of the species that occur on both transects; for example, if species X, Y, and Z are represented by 2, 4, and 8 individuals, respectively, on the first transect and 3, 5, and 7 individuals, respectively, on the second transect, pN would equal 13 (2 + 4 + 7). I also calculated the mean and standard deviation of abundance, species richness, H', and β-diversity of the transects in each of the 5 ecological sub-regions. Finally, I grouped bird species by breeding habitat and by migration status, as identified in the BBS, and calculated mean abundance and richness for each of the groups for each of the 8 ecological sub-regions.

Urban influence

First, I obtained population data for cities and towns that had >30 inhabitants in 1992 within a 300 km radius circle around the geographical center point of the Edwards Plateau Ecoregion. I used the information from the National Atlas of the United States of America (1997). Second, I calculated an index of urban influence associated with each of the midpoints of the 31 Breeding Bird Survey (BBS) transects selected in above section :

$$UI_j = \sum \frac{P_i}{D_i^2} \tag{VI.4}$$

where UI_j represents the index of urban influence at the midpoint of the jth BBS transect, P_i represents the number of inhabitants of the ith city, and D_i represents the distance (km) between the ith city and the midpoint of the jth BBS transect; UI has been used to represent the influence of cities on the market value of rural lands (Jin Shi et al 1997). Third, I calculated the mean of urban influence index value for 8 ecological sub-regions.

Spatial autocorrelation of variables

I tested for spatial autocorrelation in each of the bird diversity indexes and each of the urban influence index using the Mantel Test (Fortin and Gurevitch 1993). The Mantel Test (r) is a regression in which the variables themselves are distance, or dissimilarity (ecological distances), matrixes summarizing pair-wise similarities among sample locations:

$$r = \frac{\sum\sum stdA_{ij} * stdB_{ij}}{n-1} \text{ (sum from i to n and sum from j to n, for i} \neq \text{j)} \tag{VI.5}$$

where n is the number of sample locations, i and j identify the matrix element, B_{ij} is the Euclidian distance matrix of location points, and A_{ij} is the dissimilarity matrix of the variable of interest, the present case, avian abundance, species richness, H', β-diversity, and urban influence. For Mantel Test calculations, I used the PASSAGE program (Rosenberg 2005).

Spatial correlations between pairs of avian diversity and urban influence indexes

To determine if the indexes of avian diversity and urban influence are spatially correlated, I conducted a Cross Mantel Test (r) (Fortin and Gurevitch 1993), between each pair of avian diversity and urban influence indexes. This set of analyses tests for a spatial relationship *per se* between within-pair differences for both variables (avian diversity and urban influence), but does not indicate the degree of correlation between the values of the two variables. For example, if the differences in the values of variable A and the differences in the values of variable C increase or decrease with increasing distance (that is, with increasing geographical distance between the points where the values of the variables were measured), the variables A and C are positively spatially correlated. If the differences in the values of variable A increase with distance and the values of variable C decrease with distance, or vice versa, then variables A and C are negatively spatially correlated.

$$r = \frac{\sum\sum stdA_{ij} * stdC_{ij}}{n-1}$$ (sum from i to n and sum from j to n, for i ≠ j) (VI.6)

where n is the number of sample locations, i and j identify the matrix element, A_{ij} is the dissimilarity matrix of one of the variables of interest, that is, an index of avian diversity (species richness, avian abundance, species diversity or β-diversity), and C_{ij} is the dissimilarity matrix of the other variable of interest, that is, the urban influence index.

Correlation between values of pairs of avian diversity and urban influence indexes.

I conducted a Pearson's pair-wise correlation (ρ) between each of the avian diversity and urban influence indexes. This set of analyses identifies the degree of correlation between the values of the two variables,

$$\rho = \frac{\sum_{i=1}(u_i - m_u)(v_i - m_v)}{S_u S_v} \qquad \text{(VI.7)}$$

where u and v are two variables (u is one of the 4 avian diversity indexes and v is the urban influence index). m_u and m_v are their respective means, and S_u and S_v are their respective standard deviations. I then used the modified *t*-test for autocorrelation (CHR), which corrects the degrees of freedom based on the amount of autocorrelation in the data, to assess the correlation between each pair of spatially correlated variables (Clifford et al.1989; Dutilleul et al. 1993). The procedure calculates the amount of spatial autocorrelation of variables to determine how different the effective sample size (n') is from the number of observations.

$$n'(R) = \frac{n^2}{\sum\sum cor(u_i u_j)} \qquad \text{(VI.8)}$$

where R is the autocorrelation matrix, n is the number of observations, and u_i, and u_j are the observations of the two variables. The corrected degrees of freedom (n’-2) were then used to test the significance of the correlation; I used the PASSAGE program (Rosenberg 2005) to perform these calculations.

Results

Avian diversity indexes

Ninety-two different species, including species classified by breeding habitat as woodland, scrubland, grassland, wetland, and urban breeders, and by migration status as neotropical migrant, short-distance migrant, and permanent resident, were reported from 1990 to 1994 on the 31 BBS transects (Table A-3).

Species representing each of these breeding habitats and migration statuses were reported on virtually all transects (Table A-2). Scrubland- and urban-breeding birds were relatively more abundant and woodland, grassland, and wetland breeding birds were relatively less abundant (Table VI.1A), and migratory birds were relatively more abundant and permanent residents were relatively less abundant (Table VI.1B), in all 8 ecological sub-regions. The number of species representing these different breeding habitats and migratory statuses generally followed the same patterns as abundance, although the number of woodland-breeding species is essentially as high as the number of scrubland- and urban-breeding species (Table VI.1).

Avian abundance on the 31 transects ranged from 68 to 385 individuals (Table A-1), with higher abundances tending to occur on the northern- and southern-most transects (transect numbers 76, 66, 15, and 29, Figure VI.2). Mean abundance was highest in Mesquite Plains, followed by Brush Country, Black Land Prairie, Lampasas Cut Plains, Balcones Canyon Lands, Western Cross Timber, Mesquite Savannah and Live Oak-Mesquite Savannah, respectively (Table VI.2).

Species richness on the 31 transects ranged from 36 to 71 species (Table A-1), with higher species richness tending to occur along an east-west band of transects lying midway between the northern and southern extremes of the study area (transect numbers 140, 28, 114, and 31, Figure 4). Mean species richness was highest in Balcones Canyon Lands, followed by Brush Country, Mesquite Savannah with Live Oak-Mesquite Savannah, Mesquite Plains, Lampasas Cut Plains and Black Land Prairie having the same, relatively lower, richness (Table VI.2).

Table VI.1. *Mean (±1SD) avian abundance (total number of individuals of all species per 10 km of transect) and species richness (number of different species per 10 km of transect). Calculation is made in the 8 ecological sub-regions (Figs. 1 and 2) included in the present study, with species grouped by (A) breeding habitat and (B) migration status.*

A. Species grouped by breeding habitat.

	Woodland		Scrubland		Grassland		Wetland		Urban	
Ecological subregion	Avian abund.	Species rich.	Avian abund.	Species rich.	Avian abund.	Species rich.	Avian abund.	Species rich.	Avian abund.	Species rich.
Balcones Canyon Lands	20 (±5)	13 (±4)	51 (±13)	16 (±5)	3 (±2)	3 (±1)	12 (±2)	3 (±1)	45 (±23)	9 (±3)
Blackland Prairie	5 (±2)	6 (±2)	24 (±2)	7 (±1)	36 (±19)	3 (±1)	23 (±22)	5 (±2)	64 (±17)	8 (±1)
Brush Country	7 (±2)	6 (±3)	48 (±34)	14 (±2)	24 (±14)	3 (±1)	20 (±7)	5 (±1)	65 (±16)	8 (±1)
Lampasas Cut Plains	6 (±2)	7 (±1)	42 (±29)	7 (±2)	21 (±16)	3 (±1)	10 (±10)	4 (±2)	57 (±34)	8 (±2)
Live Oak-Mesquite Savannah	3 (±2)	4 (±2)	44 (±15)	15 (±2)	20 (±14)	2 (±1)	3 (±3)	2 (±1)	35 (±12)	5 (±2)
Mesquite Plains	4 (±2)	6 (±3)	39 (±21)	9 (±2)	24 (±12)	4 (±1)	12 (±5)	5 (±2)	83 (±37)	6 (±1)
Mesquite Savannah	7 (±4)	8 (±2)	41 (±15)	16 (±3)	10 (±6)	4 (±1)	7 (±4)	2 (±1)	58 (±10)	7 (±2)
Western Cross Timbers	5 (±1)	6 (±1)	38 (±1)	6 (±0)	9 (±2)	3 (±1)	4 (±1)	4 (±1)	49 (±1)	6 (±1)

B. Species grouped by migration status

	Neotropical migrants		Permanent residents		Short distance migrants	
Ecological subregion	Avian abundance	Species richness	Avian abundance	Species richness	Avian abundance	Species richness
Balcones Canyon Lands	47 (±19)	23 (±4)	68 (±15)	20 (±4)	50 (±14)	16 (±2)
Black Land Prairie	57 (±21)	14 (±3)	48 (±20)	11 (±1)	75 (±13)	13 (±2)
Brush Country	81 (±35)	16 (±2)	100 (±29)	17 (±3)	81 (±17)	15 (±1)
Lampasas Cut Plains	44 (±25)	16 (±3)	67 (±43)	13 (±1)	79 (±62)	14 (±3)
Live Oak-Mesquite Savannah	45 (±10)	14 (±3)	44 (±7)	13 (±1)	57 (±21)	13 (±1)
Mesquite Plains	107 (±64)	16 (±4)	68 (±34)	13 (±3)	89 (±45)	15 (±1)
Mesquite Savannah	43 (±7)	20 (±1)	62 (±1)	17 (±3)	50 (±15)	15 (±2)
Western Cross Timbers	44 (±25)	12 (±1)	59 (±0)	11 (±1)	57 (±13)	13 (±0)

Species diversity, based on values of H', on the 31 transects ranged from 1.1 to 3.6 (Table A-1), with no obvious geographical gradients within the study area (Figure 4). Mean species diversity was highest in Balcones Canyon Lands, followed by Mesquite Savannah, Lampasas Cut Plains, Live Oak-Mesquite Savannah, Black Land Prairie, Western Cross Timber, Mesquite Plains, and Brush Country (Table VI.2).

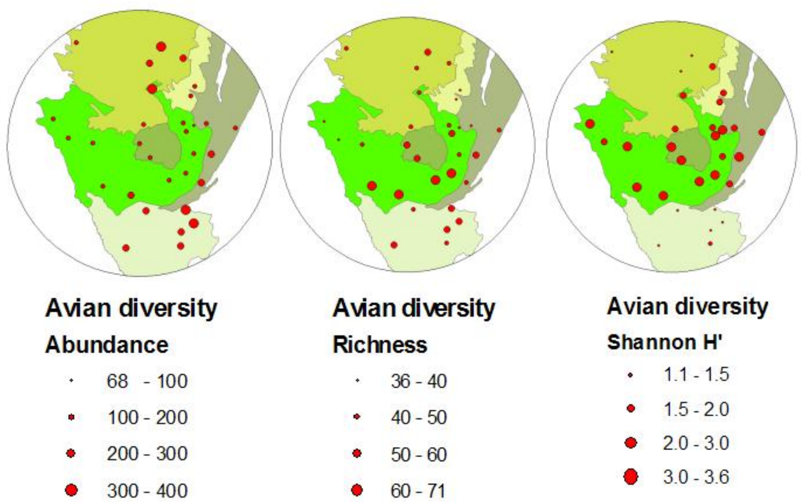

Figure VI.2. *Avian abundance (total number of individuals of all species per 10 km transect), species richness (number of different species per 10 km transect), and species diversity (H'), in the study area.*

β-diversity, based on values of 1-S', on the 31 transects ranged from 1.2 to 2.7 (Table A-1, Figure. VI.3). Mean β-diversity was lower in Brush Country, Mesquite Plains, and Balcones Canyon Lands, and higher in Live Oak-Mesquite Savannah and Lampasas Cut Plains. Highest β-diversity was founded in Live Oak Mesquite Savannah (Table VI.2).

Table VI.2. *Mean (±1SD) avian abundance (total number of individuals of all species per 10 km of transect), species richness (number of different species per 10 km of transect), species diversity (H'), and β-diversity (1-S). Calculation is made in the 8 ecological sub-regions (Fig.VI. 1) included in the present study.*

Ecological subregion	Abundance	Species richness	Species diversity	β-diversity
Balcones Canyon Lands	177(±27)	62(±9)	3,4 (±0,2)	1,6 (±0,3)
Black Land Prairie	200 (±56)	44 (±7)	2,9 (±0,1)	2,0 (±0,3)
Brush Country	269 (±38)	52 (±5)	1,4 (±0,2)	1,6 (±0,2)
Lampasas Cut Plains	194 (±131)	45 (±7)	3,1 (±0,1)	1,9 (±0,4)
Live Oak-Mesquite Savannah	147 (±38)	41 (±4)	3,1 (±0,1)	2,5 (±0,3)
Mesquite Plains	270 (±104)	47 (±5)	1,6 (±0,7)	1,7 (±0,5)
Mesquite Savannah	159 (±13)	52 (±4)	3,2 (±0,2)	1,9 (±0,2)
Western Cross Timbers	163 (±37)	38 (±1)	2,9 (±0,1)	1,8 (±0,0)

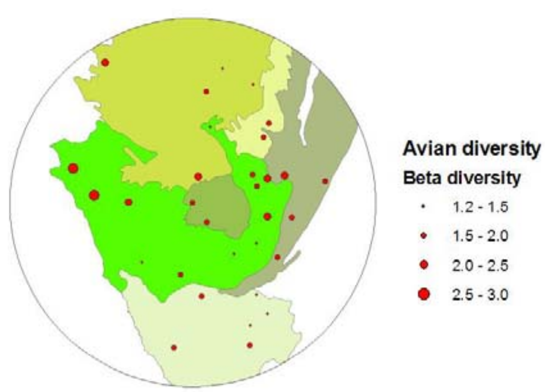

Figure VI.3. β *- diversity (1-S), based on Sorensen's similarity index (S) identified in the present study.*

Urban influence in the study area

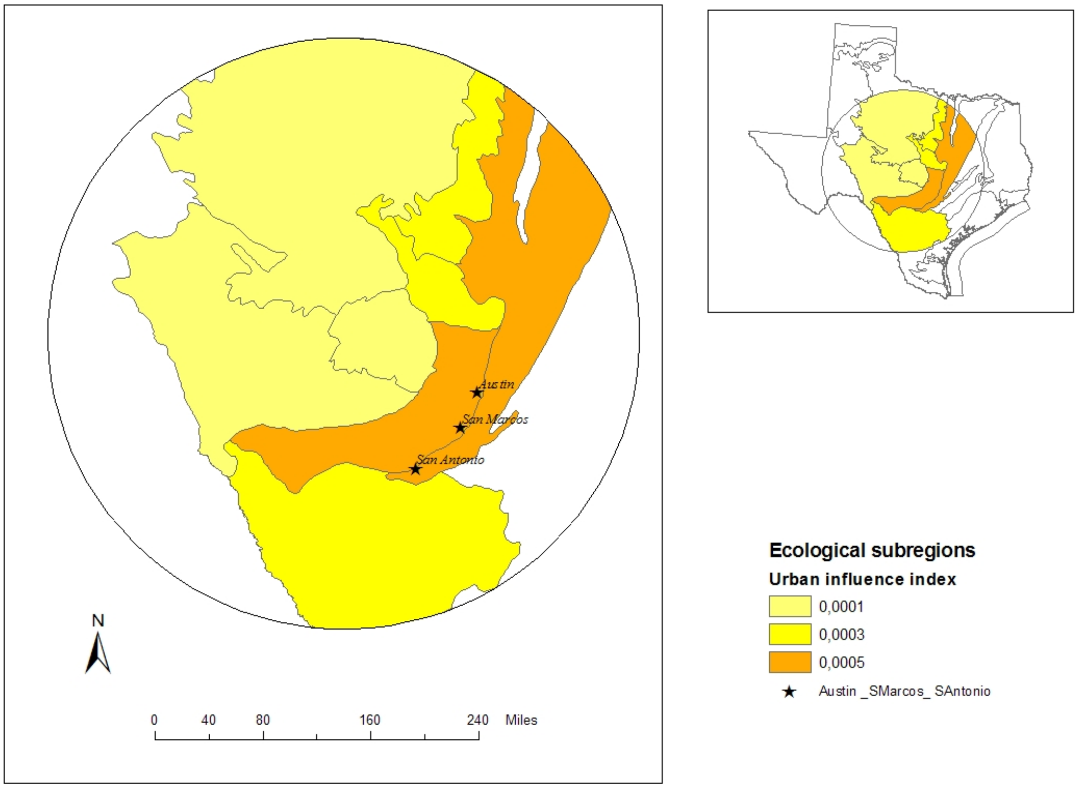

Figure VI. 4. *Urban influence in the study area by ecological subregion. Austin-San Marcos-San Antonio Metropolitan corridor location.*

Urban influence index based on values of UI_j on the 31 transects, showed a gradient east west in the study area ranged from 0.00004 (BBS transect 83092) to 0.0012 (BBS 83238), Table A-7. High values in the east of the ecoregion, are contiguous to the Austin-San Marcos-San Antonio metropolitan corridor. Figure VI.4.

At the scale of ecological subregions, the average of the urban influence index was from highest to lowest, Balcones Canyon Lands and Black Land Prairie (0,0005); Lampasas Cut plains, Brush Country, and Western Cross Timber (0.0003); and Live Oak-Mesquite Savannah, Mesquite Plains and Mesquite Savannah (0.0001), Table VI.3.

Table VI.3. *Mean, maximum, and minimum of 4 ownership property sizes (<50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) in acres, the range of sizes (maximum - minimum), and urban influence index. Calculation is given for each of the 8 ecological sub-regions included in this study.*

Ecological subregion	Data	Mean OPS	S1	S2	S3	S4	Urban influence index
Balcones Canyon Lands	Mean	1441	20	10	34	36	0.0005
	Max	4232	28	17	41	69	0.0011
	Min	298	8	2	21	17	0.0001
Blackland Prairie	Mean	387	20	16	45	20	0.0005
	Max	609	28	19	52	28	0.0008
	Min	276	12	9	38	14	0.0003
Brush Country	Mean	965	17	12	43	28	0.0003
	Max	3288	25	18	55	55	0.0008
	Min	281	10	6	29	11	0.0001
Lampasas Cut Plains	Mean	625	17	10	46	29	0.0003
	Max	636	22	11	52	31	0.0003
	Min	609	12	9	42	25	0.0002
Live Oak-Mesquite Savannah	Mean	7305	7	4	14	75	0.0001
	Max	12746	13	6	19	90	0.0001
	Min	2964	3	2	4	66	0.0001
Mesquite Plains	Mean	1530	13	9	39	40	0.0001
	Max	2391	22	13	46	56	0.0003
	Min	706	8	6	30	26	0.0000
Mesquite Savannah	Mean	1089	9	4	37	50	0.0001
	Max	1278	9	4	41	52	0.0002
	Min	995	9	4	35	46	0.0001
Western Cross Timbers	Mean	349	25	17	42	16	0.0003
	Max	355	29	19	47	18	0.0003
	Min	343	21	14	37	14	0.0003

Spatial autocorrelation of variables

The Mantel Test applied to bird diversity indexes, showed significant positive autocorrelation for Shannon index, and for β-diversity (1-Sorensen index), Urban influence index showed a positive significant autocorrelation too Table VI.4.

Table VI.4. *Degree of spatial autocorrelation of avian abundance (total number of individuals of all species per 10 km of transect), two indices* α *diversity, species richness (number of different species per 10 km of transect), species diversity (Shannon index, H'), one index of* β *-diversity y index (1-Sorensen quantitative), and urban influence index, as indicated by the Mantel's r test (Fortin and Gurevitch 1993).*

	Autocorrelation	
	Mantel's r	P
Urban influence index	0.3296	0.0081
Abundance	0.143	0.052
Richness	0.005	0.468
Relative Richness	-0.044	0.701
Shannon	0.263	0.002
β-diversity	0.201	0.012

Spatial correlation between pairs of urban influence and avian diversity indexes.

The results of the Cross Mantel Test did not show spatial correlations between urban influence index and avian diversity, Table VI.5.

Table VI.5. *Degree of spatial correlations between urban influence index and avian abundance (total number of individuals of all species per 10 km of transect), two indices α-diversity (species richness (number of different species per 10 km of transect), species diversity (Shannon index, H')), and one index of* β *-diversity index (1-Sorensen quantitative), as indicated by the Cross Mantel's r (Fortin and Gurevitch 1993).*

	Urban influence	
	Mantel's r	P
Abundance	-0.072	0.698
Richness	0.019	0.372
Shannon	-0.068	0.746
β-diversity	0.019	0.354

Correlation between pairs of urban influence and avian diversity indexes

Modified *t*-test for autocorrelation did not show significant correlations between urban influence index and avian diversity, Table VI.6

Table VI.6. *Degree of correlation between urban influence index and avian abundance (total number of individuals of all species per 10 km of transect), two indices α diversity (species richness (number of different species per 10 km of transect), species diversity (Shannon index, H')), and one index of β -diversity index (1-Sorensen quantitative), as indicated by the Modified t-Test for autocorrelation (CHR) (Clifford et al. 1989; Dutilleul et al. 1993).*

	Urban influence	
Bird diversity 1992	CRH	P
Abundance	0.050	0.872
Richness	0.134	0.665
Shannon	0.138	0.663
β-diversity	-0.155	0.587

Discussion

Urban influence did not have a significant spatial relationship in the study area in 1992. However, results of spatial correlations between the urban influence index, ownership property sizes, land structure indices, and avian diversity in previous chapters, indicate that there is an indirect relationship, through processes such as parcelization of ownership properties and habitat fragmentation.

Results of Cross Mantel and *t*-test, showed an absence of a significant spatial correlation between pairs of avian diversity indices and the urban influence index (Tables VI.5 and 6), however, high values of mean of avian species richness (66) and urban influence indices (0.00048), appeared simultaneously for example in Balcones Canyon Lands, and low values of mean of avian species richness (41) and urban influence indexes (0.00007), appeared simultaneously in Live Oak Mesquite Savannah.

As discussed in Chapter V, significant positive spatial correlation between urban influence and ownerships smaller than 500 ac and negative, with ownership sizes larger than 500 ac, in addition to the fact that Balcones Canyon Lands had only a 36% of ownership sizes larger than 500, while the Live Oak Mesquite Savannah had 75%, indicate that high urban influence appeared in same places with high land "parcelization" (Alig et al. 2004).

As discussed in Chapter IV, significant negative spatial correlation were identified among ownership properties smaller than 500ac, and landscape structure indices as percent of land PL, mean patch size MN and Proximity PROX. This means that land parcelization produces a decrease in land cover of natural vegetation (woodland, scrubland, grassland), in mean patch size, and proximity among patches.

In Black Land Prairie where the average ownership property size was 387 ac, woodland habitat had a higher grade of habitat fragmentation (indicated by mean patch size MN 2.5 ac and low proximity (1336) among patches Balcones Canyon Lands had an average of ownership property size, more than double (1441 ac.) and mean patch sizes was 7.8 ac, three times than in Black Land Prairies, and proximity among patches was 50323 (Chapter IV).

Those physical and ecological characteristics gave Balcones Canyon Lands an avian species richness of 62 species, 44 for Black Land Prairies, and 41 for Live Oak-Mesquite Savannah. Species richness increased with urban influence until ownership property size reached 500 ac. This limit corresponds to an “intermediate urban development” in which there is an increase in landscape heterogeneity, because of the establishment of ornamental vegetation patches, the increase of between-habitat borders, and higher water availability that provides new resources for the birds (Blair R., 1996; McKinney 2002). In those conditions, as soon as less tolerant to urbanization species gets extinguished, new species adapted to urban areas enter the regional pool of species. This is the case of Balcones Canyon Lands, which had the highest number (9) of urban species (Chapter III).

As soon as urban influence increased over the threshold and ownership size decreases below 500 ac, increasing habitat fragmentation, then more species get extinguished, and only urban species remain as part of species pool. In this case, species richness decreases, as in the case of Black Land Prairies and Western Cross Timber subregions.

This seems to follow the findings of Luck and Wu (2002), who stated “urbanization produces habitat fragmentation, inducing the extinction of species adapted to that habitat and facilitating the invasion of ubiquitous ones”. The presence of birds, such as the Chipping sparrow, Rock dove, Blue Jay, Mourning dove and Northern Mockingbird in ecological subregions located towards the eastern side of the Edwards Plateau, could be a sign of the disappearance of some woodland species, which would have led to the entrance of these urban-adapted species.

Finally it is possible to say that urban influence did not have a direct relationship with avian diversity. It is indirect through ownership parcelization and habitat fragmentation. This is especially important in the ecological subregions located in the eastern portion of the Edwards Plateau, which is an area with a high level of endemism (BirdLife International 2003), and in areas like this there is not a difference between a local and global extinction.

Conclusions

Alpha and beta diversity indices are spatially autocorrelated variables in the study area, indicating that there were driving ecological and environmental factors that had produced a spatial pattern of the avian diversity. Although diversity indices did not have a significant spatial correlation with the urban influence index in the study area, spatial correlations between avian diversity, landscape structure, ownership property size, and urban influence in previous chapters, indicate that there is an indirect relationship through ownership properties parcelization and habitat fragmentation. In the study area, avian species richness increased when urban influence increased, until a threshold in which urban influence pushed ownership properties to reach sizes lesser than 500 ac. This threshold probably corresponded to an “intermediate urban development”. In 1992 Balcones Canyon Lands subregion had reached an intermediate degree of urban development. There were found the highest values of species richness (62) and Shannon diversity H’ (3.4), but urban birds could have been displacing native bird species, as indicated by the highest values of urban bird richness, found in this subregion, Table VI.

CHAPTER VII

GENERAL CONCLUSIONS

This study showed a complex system of interaction among urban influence, ownership property size, landscape structure and avian diversity. Significant spatial correlations between avian diversity and landscape structure; between landscape structure and ownership property size; and between property size and urban influence, indicate that there is an indirect relationship between urban influence and avian diversity.

Results indicated that as urban influence increases, ownership property size declines. At 500 ac, average property size can be said to reflect “intermediate urban development”. At this level of urbanization, there began a significant negative correlation between urban influence and property size. Simultaneously, landscape structure at class level (mean patch size MN and proximity Index PROX) is affected by ownership property sizes. The spatial analysis made it possible to identify 500 ac ownership property size as the “threshold of habitat fragmentation”. Patch size of scrubland habitat for example, decreases when land becomes dominated by ownership property sizes smaller than 500 ac, and distance among habitat patches decreases when land becomes dominated by ownership property sizes smaller than 100 ac.

Spatial analysis made possible to identify too, that avian α -diversity (richness) is affected by landscape structure at class level; it increases with mean patch size MN and proximity index PROX. Those landscape metrics are directly related to availability of suitable habitat for avian meta-populations. Isolated patches act as habitat islands, which colonization or repopulation depends of patch area and distance among habitat patches. This is especially important for conservation of suitable habitat for native permanent resident as Northern Bobwhite and short-distance native migrants as Field Sparrow.

At the scale of ecological subregion, was possible to prioritize the Balcones Canyon Land as an ecological subregion for conservation and the Lampasas Cut Plains for ecological restoration with respect to avian α -diversity. In addition, it was possible to prioritize the conservation of landscape diversity in the Live Oak-Mesquite for native avian species turnover (β -diversity). Although Balcones Canyon Lands and Lampasas Cut Plains showed high percentages of land covered by farms smaller than 500 ac (74% and 64% respectively), the former had an average ownership property size above the threshold of fragmentation (1440 ac) and the latter an average ownership property size below the threshold of fragmentation (500 ac).

In Balcones Canyon Lands, urban influence had enriched avian diversity in 1992, adding 13 species of urban birds; so, values of avian α -diversity were elevated. However, in the contiguous subregions Lampasas Cut Plains and Black Land Prairies, the urbanization level had pushed ownership properties to reach sizes below the habitat fragmentation threshold (500 ac).

There, urban bird species became dominant and avian α-diversity decreased because of the loss of native bird species.

In the Live Oak-Mesquite Savannah the high land mosaic diversity provided suitable environment for native avian species turnover. This subregion showed the highest average ownership property size value (7305 ac.), and the highest values of patch richness and β-diversity. In this subregion, there is still opportunity to conserve patch richness that supports high avian diversity.

Finally, it was possible to prioritize counties for habitat conservation or habitat restoration. Counties that had an average ownership property size close to the threshold of 500 ac were Kendall (528 ac) and Hays (658 ac) in Balcones Canyon Lands; Coryell (609 ac) and Lampasas (628 ac), in Lampasas Cut Plains; and, Palo Pinto (706 ac) in Mesquite Plains. For these counties, it is especially important that conservation plans be focused on monitoring the parcelization process and on habitat conservation. Karnes (364 ac) and Wilson (281 ac) in Brush Country; Caldwell (276 ac) and Falls (366 ac) in Black Land Prairies; and Hood (343 ac.) in Mesquite Plains, had the average ownership property sizes, below of the threshold of fragmentation. For these counties, the emphasis needs to be on native habitat restoration.

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APPENDIX 1

Table A.1. *Avian abundance (total number of individuals of all species per 10 km of transect) and two indices α-diversity, species richness (number of different species per 10 km of transect), species diversity (Shannon index, H'), and β-diversity (1-Sorensen). Calculation is made for the 31 North American Breeding Bird Survey (BBS) transects from which data were drawn for the present study. The ecoregion and ecological sub-region (Figs. 1 and 2) to which each transect belongs are also presented.*

BBS transect code	Ecoregion	Ecological subregion	Abundance	Species richness	Species diversity H'	β-diversity (1-Sorensen)
83015	South Texas Brush	Brush Country	311	57	1.5	1.4
83016	South Texas Brush	Brush Country	281	47	1.6	1.7
83017	South Texas Brush	Brush Country	220	51	1.3	1.5
83018	South Texas Brush	Brush Country	274	53	1.1	1.9
83027	Blackland Prairie	Blackland Prairie	261	44	2.9	2.0
83028	Edwards Plateau	Balcones Canyon Lands	189	64	3.5	1.5
83029	South Texas Brush	Brush Country	304	59	1.5	1.3
83030	South Texas Brush	Brush Country	226	46	1.2	1.7
83031	Edwards Plateau	Balcones Canyon Lands	218	63	3.4	1.6
83042	Llano Uplift	Mesquite Savannah	144	55	3.3	1.8
83043	Llano Uplift	Mesquite Savannah	170	54	3.2	1.9
83048	Blackland Prairie	Blackland Prairie	173	45	2.9	1.9
83050	Edwards Plateau	Lampasas Cut Plains	136	36	2.8	2.4
83051	Edwards Plateau	Lampasas Cut Plains	68	36	3.1	2.4
83052	Edwards Plateau	Lampasas Cut Plains	148	54	3.2	1.6
83053	Edwards Plateau	Lampasas Cut Plains	181	44	3.0	2.0
83054	Llano Uplift	Mesquite Savannah	162	48	3.0	2.1
83062	Oak Woods	Western Cross Timbers	189	38	2.8	1.8
83064	Oak Woods	Western Cross Timbers	136	37	2.9	1.8
83065	Rolling Plains	Mesquite Plains	286	45	2.7	1.5
83066	Edwards Plateau	Lampasas Cut Plains	377	47	2.9	1.4
83067	Rolling Plains	Mesquite Plains	274	43	1.2	2.0
83076	Rolling Plains	Mesquite Plains	385	55	1.2	1.2
83086	Edwards Plateau	Live Oak-Mesquite Savannah	164	38	3.1	2.7
83092	Rolling Plains	Mesquite Plains	133	45	1.3	2.2
83112	Edwards Plateau	Live Oak-Mesquite Savannah	104	39	3.0	2.6
83113	Edwards Plateau	Live Oak-Mesquite Savannah	174	46	3.2	2.2
83114	Edwards Plateau	Balcones Canyon Lands	158	65	3.5	1.5
83139	Blackland Prairie	Blackland Prairie	231	52	3.1	1.7
83140	Edwards Plateau	Balcones Canyon Lands	149	71	3.6	1.4
83238	Edwards Plateau	Balcones Canyon Lands	173	46	3.0	2.1

Table A.2. *Avian abundance (total number of individuals of all species per 10 km of transect) and species richness (number of different species per 10 km of transect). Calculation is made for each of the 31 North American Breeding Bird Survey (BBS) transects from which data were drawn for the present study, with species grouped by breeding habitat (woodland, scrubland, grassland, wetland, urban) and migration status (neotropical migrant, short distance migrant, permanent resident).*

			Woodland		Scrubland		Grassland		Wetland		Urban		Neotropical migrants		Short distance migrants		Permanent residents	
BBS transect code	Ecoregion	Ecological subregion	Avian abundance	Species rich	Avian abundance	Species rich	Avian abundance	Species rich	Avian abundance	Species rich	Avian abundance	Species rich	Avian abundance	Species rich	Avian abundance	Species rich	Avian abundance	Species rich
83015	South Texas Brush	Brush Country	7	10	50	14	38	4	16	4	85	7	108	17	94	15	104	21
83016	South Texas Brush	Brush Country	4	1	20	13	39	4	22	7	73	7	124	16	91	13	60	12
83017	South Texas Brush	Brush Country	9	5	40	14	33	4	20	5	61	8	62	14	76	15	76	17
83018	South Texas Brush	Brush Country	8	4	113	17	16	2	10	5	39	6	34	12	99	15	129	20
83027	Blackland Prairie	Blackland Prairie	4	5	22	7	27	2	56	6	83	7	70	13	91	13	54	11
83028	Edwards Plateau	Balcones Canyon Lands	23	14	47	14	3	3	11	5	47	9	45	22	59	18	74	19
83029	South Texas Brush	Brush Country	10	9	24	13	8	2	19	6	56	9	99	19	54	16	134	19
83030	South Texas Brush	Brush Country	5	4	38	11	10	3	32	4	76	8	56	15	74	13	95	15
83031	Edwards Plateau	Balcones Canyon Lands	20	12	64	19	6	4	14	2	45	9	78	24	64	15	63	23
83042	Llano Uplift	Mesquite Savannah	10	9	50	18	4	3	4	2	54	8	42	19	35	16	62	19
83043	Llano Uplift	Mesquite Savannah	7	9	50	17	10	3	6	1	51	5	37	20	65	15	62	18
83048	Blackland Prairie	Blackland Prairie	5	6	23	7	48	2	16	5	46	9	61	13	61	14	33	11
83050	Edwards Plateau	Lampasas Cut Plains	4	4	24	6	14	3	10	3	52	7	27	12	69	10	31	10

Table A.2 Continued

			Woodland		Scrubland		Grassland		Wetland		Urban		Neotropical migrants		Short distance migrants		Permanent residents	
83051	Edwards Plateau	Lampasas Cut Plains	5	5	19	6	7	2	3	2	20	6	17	11	18	11	30	12
83052	Edwards Plateau	Lampasas Cut Plains	5	7	36	10	7	3	5	6	46	10	40	18	53	17	51	14
83053	Edwards Plateau	Lampasas Cut Plains	5	8	28	6	33	3	8	3	61	9	40	16	80	14	57	12
83054	Llano Uplift	Mesquite Savannah	3	5	24	13	16	5	11	3	70	8	50	20	50	13	61	13
83062	Oak Woods	Western Cross Timbers	4	5	38	6	10	2	4	4	50	5	62	12	66	13	59	10
83064	Oak Woods	Western Cross Timbers	6	7	37	6	7	3	3	3	48	7	26	11	48	13	59	11
83065	Rolling Plains	Mesquite Plains	6	7	42	8	19	3	16	5	114	6	79	14	87	14	106	13
83066	Edwards Plateau	Lampasas Cut Plains	9	6	83	7	37	3	25	4	101	7	78	17	164	13	128	12
83067	Rolling Plains	Mesquite Plains	3	4	54	11	33	4	16	3	104	5	68	13	137	15	65	13
83076	Rolling Plains	Mesquite Plains	4	9	51	10	35	5	12	4	82	6	203	21	101	15	77	15
83086	Edwards Plateau	Live Oak-Mesquite Savannah	1	2	46	13	36	3	2	2	39	6	45	12	68	13	50	12
83092	Rolling Plains	Mesquite Plains	1	2	9	7	9	4	5	7	30	8	77	16	29	14	24	9
83112	Edwards Plateau	Live Oak-Mesquite Savannah	4	5	29	15	9	1	0	1	22	3	35	13	33	12	37	13
83113	Edwards Plateau	Live Oak-Mesquite Savannah	5	4	58	16	15	2	6	2	45	6	54	18	69	13	46	14
83114	Edwards Plateau	Balcones Canyon Lands	20	15	67	23	0	1	13	1	13	5	47	27	51	17	47	20

Table A.2 Continued

			Woodland		Scrubland		Grassland		Wetland		Urban		Neotropical migrants		Short distance migrants		Permanent residents	
83139	Black Land Prairie	Black Land Prairie	8	8	27	7	54	3	11	7	73	8	71	18	77	14	74	11
83140	Edwards Plateau	Balcones Canyon Lands	24	16	39	16	1	3	12	3	41	12	36	26	27	18	69	23
83238	Edwards Plateau	Balcones Canyon Lands	11	7	40	9	4	2	8	3	79	10	30	16	48	14	87	13

Table A.3. *Avian species reported from 1990 to 1994 on the 31 North American Breeding Bird Survey (BBS) transects from which data were drawn for the present study.*

Breeding Habitat	Migration Status	Species
Woodland	Neotropical migrant	Black-and-white Warbler
Woodland	Neotropical migrant	Blue-gray Gnatcatcher
Woodland	Neotropical migrant	Great Crested Flycatcher
Woodland	Neotropical migrant	Red-eyed Vireo
Woodland	Neotropical migrant	Summer Tanager
Woodland	Neotropical migrant	Vermilion Flycatcher
Woodland	Neotropical migrant	Yellow-throated Vireo
Woodland	Neotropical migrant	Yellow-throated Warbler
Woodland	Neotropical migrant	Black-chinned Hummingbird
Woodland	Neotropical migrant	Ruby-throated Hummingbird
Woodland	Neotropical migrant	Black-bellied Whistling-Duck
Woodland	Neotropical migrant	Chuck-will's-widow
Woodland	Permanent resident	Barred Owl
Woodland	Permanent resident	Eastern Screech-Owl
Woodland	Permanent resident	Eastern Wood-Pewee
Woodland	Permanent resident	Carolina Chickadee
Woodland	Permanent resident	Golden-fronted Woodpecker
Woodland	Permanent resident	Ladder-backed Woodpecker
Woodland	Permanent resident	Black-crested Titmouse
Woodland	Permanent resident	Red-bellied Woodpecker
Woodland	Permanent resident	Wild Turkey
Woodland	Short distance migrant	Cooper's Hawk
Woodland	Short distance migrant	Red-shouldered Hawk
Woodland	Short distance migrant	Brown-crested Flycatcher
Scrubland	Neotropical migrant	Ash-throated Flycatcher
Scrubland	Neotropical migrant	Bell's Vireo
Scrubland	Neotropical migrant	Blue Grosbeak
Scrubland	Neotropical migrant	Golden-cheeked Warbler
Scrubland	Neotropical migrant	Gray Vireo
Scrubland	Neotropical migrant	Indigo Bunting
Scrubland	Neotropical migrant	Lesser Nighthawk
Scrubland	Neotropical migrant	White-eyed Vireo
Scrubland	Neotropical migrant	Black-capped Vireo
Scrubland	Neotropical migrant	Lark Sparrow
Scrubland	Neotropical migrant	Scott's Oriole
Scrubland	Neotropical migrant	Painted Bunting
Scrubland	Permanent resident	Greater Roadrunner
Scrubland	Permanent resident	Harris's Hawk
Scrubland	Permanent resident	Bushtit
Scrubland	Permanent resident	Cactus Wren
Scrubland	Permanent resident	Carolina Wren
Scrubland	Permanent resident	Common Pauraque

Table A.3 Continued

Breeding Habitat	Migration Status	Species
Scrubland	Permanent resident	Long-billed Thrasher
Scrubland	Permanent resident	Olive Sparrow
Scrubland	Permanent resident	Verdin
Scrubland	Permanent resident	Canyon Towhee
Scrubland	Permanent resident	Cardinal/Pyrrhuloxia
Scrubland	Permanent resident	Common Ground-Dove
Scrubland	Permanent resident	Northern Bobwhite
Scrubland	Permanent resident	Northern Cardinal
Scrubland	Permanent resident	Pyrrhuloxia
Scrubland	Permanent resident	Rufous-crowned Sparrow
Scrubland	Permanent resident	Scaled Quail
Scrubland	Permanent resident	Western Scrub-Jay
Scrubland	Short distance migrant	Common Poorwill
Scrubland	Short distance migrant	Black-throated Sparrow
Scrubland	Short distance migrant	Groove-billed Ani
Scrubland	Short distance migrant	Curve-billed Thrasher
Scrubland	Short distance migrant	Field Sparrow
Scrubland	Short distance migrant	Lesser Goldfinch
Grassland	Neotropical migrant	Burrowing Owl
Grassland	Neotropical migrant	Dickcissel
Grassland	Neotropical migrant	Grasshopper Sparrow
Grassland	Short distance migrant	Eastern Meadowlark
Grassland	Short distance migrant	Cassin's Sparrow
Grassland	Short distance migrant	Horned Lark
Wetland-open water	N/A	American Avocet
Wetland-open water	N/A	Belted Kingfisher
Wetland-open water	N/A	Black-necked Stilt
Wetland-open water	N/A	Great Blue Heron
Wetland-open water	N/A	Great Egret
Wetland-open water	N/A	Green Heron
Wetland-open water	N/A	Pied-billed Grebe
Wetland-open water	N/A	Bewick's Wren
Wetland-open water	N/A	Cattle Egret
Wetland-open water	N/A	American Coot
Wetland-open water	N/A	Common Moorhen
Wetland-open water	N/A	Northern Pintail
Wetland-open water	N/A	Northern Shoveler
Wetland-open water	Short distance migrant	Red-winged Blackbird
Urban	Neotropical migrant	Chimney Swift
Urban	Neotropical migrant	Purple Martin
Urban	Neotropical migrant	Chipping Sparrow
Urban	Permanent resident	Northern Mockingbird
Urban	Permanent resident	House Sparrow
Urban	Permanent resident	Inca Dove
Urban	Permanent resident	Rock Dove

Table A.3 Continued

Breeding Habitat	Migration Status	Species
Urban	Short distance migrant	European Starling
Urban	Short distance migrant	Blue Jay
Urban	Short distance migrant	Common Grackle
Urban	Short distance migrant	House Finch
Urban	Short distance migrant	Mourning Dove

Table A.4. *The 5 land cover classes used in this study and their relation to the 21 land cover classes identified in the National Land Cover Data (NLCD). I first reclassified the 21 NLCD classes into 10 classes, as described in the text, and then identified those classes that provide habitat for the bird species included in the present study.*

NLCD Code	NLCD Land Cover Class	Reclassification code	Land cover reclassification	Classes providing habitat for birds in this study
11	Water	1	Water	
12	Water	1	Water	
21	Developed	2	Urban	Urban
22	Developed	2	Urban	
23	Developed	2	Urban	
31	Barren	3	Barren	
32	Barren	3	Barren	
33	Barren	3	Barren	
41	Forest upland	4	Woodland	Woodland
42	Forest upland	4	Woodland	
43	Forest upland	4	Woodland	
51	Shrubland	5	Shrubland	Shrubland
61	Non-Natural Woody	6	Non-Natural Woody	
71	Herbaceous Upland Natural/Semi-Natural Vegetation	7	Grassland	Grassland
81	Herbaceous planted/Cultivated	8	Pasture	
82	Herbaceous planted/Cultivated	9	Herbaceous planted/Cultivated	
83	Herbaceous planted/Cultivated	9	Herbaceous planted/Cultivated	
84	Herbaceous planted/Cultivated	9	Herbaceous planted/Cultivated	
85	Herbaceous planted/Cultivated	2	Urban	Urban
91	Wetland	10	Wetland	Wetland
92	Wetland	10	Wetland	

Table A.5. *Landscape structure indexes at the landscape mosaic level (richness of patches, PR; and Shannon Diversity, SHDI) and the land cover class level (percent of land, PL; patch density, PD (number of patches / 100 ha); mean patch size, MN (ha); proximity index, (PROX). Calculation is made for the buffer scenes around the 31 North American Breeding Bird Survey (BBS) transects from which data were drawn for the present study.*

			Landscape mosaic indices						Land cover class indexes															
					Woodland				Scrubland				Grassland				Wetland				Urban			
BBS transect code	Ecoregion	Ecologlogical subregion	PR	SHDI	PL	PD	MN	PROX	PL	PD	MN	PROX	PL	PD	MN	PROX	PL	PD	MN	PROX	PL	PD	MN	PROX
83015	South Texas Brush	Brush Country	9	1.7	17	11	2	166	24	17	1	222	16	15	1	68	1.1	2.2	0.5	3	1	0.3	3	37
83016	South Texas Brush	Brush Country	9	1.4	13	11	1	88	55	5	11	45153	8	6	1	22	0.6	0.8	0.7	13	1	0.4	1	9
83017	South Texas Brush	Brush Country	9	1.5	16	13	1	74	48	8	6	52866	8	6	1	18	0.9	1.5	0.6	7	0	0.1	2	6
83018	South Texas Brush	Live Oak-Mesquite Savannah	10	1.2	3	4	1	88	60	6	11	1E+05	24	11	2	690	0.2	0.2	0.9	9	0	0.1	1	2
83027	Blackland Prairie	Lampasas Cut Plains	9	1.7	27	8	3	3412	12	18	1	9	23	10	2	103	0.0	0.2	0.1	0	2	0.5	5	118
83028	Edwards Plateau	Balcones Canyon Lands	9	1.2	46	9	5	####	7	26	0	30	40	8	5	4372	0.0	0.2	0.1	0	2	0.5	5	43
83029	South Texas Brush	Brush Country	9	1.7	16	10	2	405	22	15	1	70	14	12	1	27	0.2	0.9	0.3	1	5	0.5	11	2160
83030	South Texas Brush	Brush Country	10	1.5	14	10	1	597	39	7	5	7799	6	6	1	5	0.1	0.2	0.2	0	1	0.2	2	24
83031	Edwards Plateau	Balcones Canyon Lands	10	1.0	69	3	20	1E+05	13	10	1	81	8	8	1	14	0.0	0.1	0.1	0	0	0.2	1	4
83042	Llano Uplift	Live Oak-Mesquite Savannah	9	1.2	37	12	3	3160	43	9	5	31159	16	10	2	80	0.0	0.1	0.1	0	0	0.2	2	7
83043	Llano Uplift	Live Oak-Mesquite Savannah	9	1.2	15	14	1	78	53	6	9	84367	25	9	3	847	0.0	0.1	0.2	0	1	0.3	3	15
83048	Blackland Prairie	Lampasas Cut Plains	10	1.7	24	8	3	621	16	15	1	40	12	14	1	21	0.2	0.7	0.3	1	0	0.4	1	7
83050	Edwards Plateau	Lampasas Cut Plains	9	1.7	29	9	3	1039	13	16	1	17	26	8	4	886	0.1	0.6	0.1	0	3	0.5	6	89
83051	Edwards Plateau	Lampasas Cut Plains	9	1.4	23	10	2	435	19	16	1	453	46	6	8	8356	0.0	0.3	0.1	0	4	0.2	15	511

Table A.5 Continued

			Landscape mosaic indices								Land			cover class			indexes							
					Woodland				Scrubland				Grassland				Wetland				Urban			
83052	Edwards Plateau	Lampasas Cut Plains	9	1.3	19	10	2	303	37	13	3	3957	35	10	4	1892	0.0	0.2	0.1	0	2	0.2	8	165
83053	Edwards Plateau	Lampasas Cut Plains	10	1.3	13	12	1	62	46	9	5	44837	30	11	3	269	0.0	0.1	0.1	0	0	0.3	2	12
83054	Llano Uplift	Live Oak-Mesquite Savannah	9	1.3	6	10	1	94	43	5	8	45072	20	10	2	174	0.0	0.2	0.1	0	1	0.3	2	26
83062	Oak Woods	Mesquite Plains	10	1.5	25	8	3	219	12	14	1	588	44	8	6	20418	0.1	0.5	0.1	0	2	1.0	2	16
83064	Oak Woods	Mesquite Plains	10	1.4	18	11	2	100	43	9	5	14981	26	13	2	606	0.0	0.3	0.1	0	1	0.8	2	41
83065	Rolling Plains	Mesquite Plains	9	1.6	28	7	4	847	14	19	1	13	33	8	4	3911	0.1	0.1	1.2	35	1	0.3	2	56
83066	Edwards Plateau	Mesquite Plains	10	1.4	13	10	1	29	40	9	5	29115	32	11	3	2104	0.0	0.1	0.2	0	1	0.9	1	7
83067	Rolling Plains	Lampasas Cut Plains	9	1.1	1	3	1	4	21	12	2	270	64	5	12	138471	0.1	0.6	0.2	0	0	0.1	2	41
83076	Rolling Plains	Mesquite Plains	9	1.5	4	6	1	11	14	14	1	52	43	8	6	28348	0.1	0.5	0.2	0	0	0.0	9	95
83086	Edwards Plateau	Live Oak-Mesquite Savannah	10	0.9	0	0	0	0	52	9	6	73598	42	9	4	18424	0.0	0.0	0.5	1	0	0.7	0	3
83092	Rolling Plains	Live Oak-Mesquite Savannah	9	1.2	0	0	0	0	14	17	1	207	40	4	9	103102	0.1	0.2	0.4	1	0	0.2	1	20
83112	Edwards Plateau	Live Oak-Mesquite Savannah	8	0.7	1	1	1	4	64	6	12	3E+05	35	14	3	3426	0.0	0.0	0.1	0	0	0.5	0	2
83113	Edwards Plateau	Live Oak-Mesquite Savannah	10	0.5	1	2	1	2	87	1	128	5E+05	10	18	1	8	0.0	0.0	0.1	0	0	0.3	1	12
83114	Edwards Plateau	Balcones Canyon Lands	9	1.0	15	7	2	165	64	3	20	2E+05	20	10	2	394	0.0	0.0	0.1	0	0	0.1	3	20
83139	Oak Woods	Lampasas Cut Plains	10	1.6	13	9	1	272	13	16	1	40	19	12	2	172	0.0	0.2	0.1	0	2	0.4	5	91
83140	Edwards Plateau	Balcones Canyon Lands	9	1.1	56	7	8	####	8	25	0	22	31	10	3	360	0.0	0.2	0.1	0	3	0.4	7	72
83238	Edwards Plateau	Balcones Canyon Lands	9	1.4	40	9	4	8122	14	21	1	78	36	9	4	5843	0.1	0.3	0.2	0	6	0.6	9	1265

Table A.6. *Mean (±1SD) landscape structure indexes at the landscape mosaic level (richness of patches, PR; and Shannon Diversity, SHDI) and the land cover class level (percent of land, PL; patch density, PD (number of patches / 100 ha); mean patch size, MN (ha); proximity index, PROX). Calculation is made for the buffer scenes in each of the 5 ecological sub-regions.*

	Land mosaic level		Class level Grassland				Class level Scrubland				Class level Urban				Class level Wetland				Class level Woodland			
Ecological subregions	PR	SHDI	MN	PD	PL	PROX	MN	PD	PL	PROX	MN	PD	PL	PROX	MN	PD	PL	PROX	MN	PD	PL	PROX
Balcones Canyon Lands	9,2 (0,4)	1,1 (0,2)	3,0 (,6)	9,0 (1,0)	27 (13)	2196,6 (2711,8)	4,4 (8,7)	17,0 (10,1)	21,2 (24)	47629,6 (106385)	5,0 (3,2)	0,4 (0,2)	2,2 (2,5)	280,8 (550,8)	0,1 (0)	0,2 (0,1)	0 (0)	0 (0)	7,8 (7,2)	7,0 (2,4)	45,2 (20,1)	50323 (57297)
Blackland Prairie	9,5 (0,6)	1,7 (0,1)	2,3 (1,3)	11,0 (2,6)	20 (6,1)	295,5 (398,5)	1 (0)	16,3 (1,3)	13,5 (1,7)	26,5 (15,9)	4,3 (2,2)	0,5 (0,1)	1,8 (1,3)	76,3 (48)	0,2 (0,1)	0,4 (0,3)	0,1 (0,1)	0,3 (0,5)	2,5 (1)	8,5 (0,6)	23,3 (7,1)	1336 (1419)
Brush Country	9,3 (0,5)	1,5 (0,2)	1,2 (0,4)	9,3 (3,9)	12,7 (6,8)	138,3 (271,1)	5,8 (4,5)	9,7 (5)	41,3 (16)	37505,3 (46037)	3,3 (3,8)	0,3 (0,2)	1,3 (1,9)	373 (875,5)	0,5 (0,3)	1,0 (0,8)	0,5 (0,4)	5,5 (5)	1,3 (0,5)	9,8 (3,1)	13,2 (5,2)	236,3 (216,3)
Lampasas Cut Plains	9,5 (0,6)	1,4 (0,1)	4,5 (2,4)	9,5 (2,4)	35,8 (7,1)	3155,3 (3562,7)	3,5 (1,9)	11,8 (3,4)	35,5 (11,6)	19590,5 (21124,6)	6,5 (6,5)	0,4 (0,3)	1,8 (1,7)	173,8 (236,5)	0,1 (0,1)	0,2 (0,1)	0 (0)	0 (0)	1,5 (0,6)	10,5 (1)	17 (4,9)	207,3 (194,9)
Live Oak-Mesquite Savannah	9,3 (1,2)	0,7 (0,2)	2,7 (1,5)	13,7 (4,5)	29 (16,8)	7286 (9796)	48,7 (68,8)	5,3 (4)	67,7 (17,8)	301947,7 (215326)	0,3 (0,6)	0,5 (0,2)	0 (0)	5,7 (5,5)	0,2 (0,2)	0 (0)	0 (0)	0,3 (0,6)	0,7 (0,6)	1 (0)	0,7 (0,6)	2 (2)
Mesquite Plains	9 (0)	1,4 (0,2)	7,8 (3,5)	6,3 (2,1)	45 (13,3)	68458 (62921)	1,3 (0,5)	15,5 (3,1)	15,8 (3,5)	135,5 (122,7)	3,5 (3,7)	0,2 (0,1)	0,3 (0,5)	53 (31,7)	0,5 (0,5)	0,4 (0,2)	0,1 (0)	9,0 (17,3)	1,5 (1,7)	4,0 (3,2)	8,3 (13,3)	215,5 (421)
Mesquite Savannah	9 (0)	1,2 (0,1)	2,3 (0,6)	9,7 (0,6)	20,3 (4,5)	367 (418,3)	7,3 (2,1)	6,7 (2,1)	46,3 (5,8)	53532,7 (27594,6)	2,3 (0,6)	0,3 (0,1)	0,7 (0,6)	16 (9,5)	0,1 (0,1)	0,1 (0,1)	0 (0)	0 (0)	1,7 (1,2)	12 (2)	19,3 (15,9)	1110,7 (1774,8)
Western Cross Timbers	10 (0)	1,5 (0,1)	4,0 (2,8)	10,5 (3,5)	35 (12,7)	10512 (14009,2)	3,0 (2,8)	11,5 (3,5)	27,5 (21,9)	7784,5 (10177)	2 (0)	0,9 (0,1)	1,5 (0,7)	28,5 (17,7)	0,1 (0)	0,4 (0,1)	0,1 (0,1)	0 (0)	2,5 (0,7)	9,5 (2,1)	21,5 (4,9)	159,5 (84,1)

Table A.7. *Mean of rural (farm and ranch) ownership property size (USDA 1992), the proportions of rural acreage in each of 4 size classes: <50 (S1), 50-99 (S2), 100-500 (S3), >500 (S4) in acres, and the urban influence index. Calculation is made in the 27 Texas counties associated with the 31 North American Breeding Bird Survey (BBS) transects described in Chapter II. The ecoregion and ecological sub-region (Figure 1) to which each transect belongs also are presented.*

BBS transect code	Ecoregion	Ecological subregion	County	Mean OPS	S1	S2	S3	S4	Urban influence index
83015	South Texas Brush	Brush Country	Karnes	364	12	13	55	19	0.00030
83016	South Texas Brush	Brush Country	Live Oak	790	13	8	46	34	0.00016
83017	South Texas Brush	Brush Country	Atascosa	618	20	14	41	25	0.00024
83018	South Texas Brush	Live Oak-Mesquite Savannah	Dimmitt	3288	10	6	29	55	0.00009
83027	Blackland Prairie	Lampasas Cut Plains	Caldwell	276	22	17	47	14	0.00084
83028	Edwards Plateau	Balcones Canyon Lands	Kendall	528	22	10	41	27	0.00043
83029	South Texas Brush	Brush Country	Wilson	281	25	18	45	11	0.00080
83030	South Texas Brush	Brush Country	Medina	451	23	13	40	25	0.00031
83031	Edwards Plateau	Balcones Canyon Lands	Uvalde	1487	13	8	31	48	0.00015
83042	Llano Uplift	Live Oak-Mesquite Savannah	Mason	995	9	4	35	52	0.00015
83043	Llano Uplift	Live Oak-Mesquite Savannah	Mason	995	9	4	35	52	0.00011
83048	Blackland Prairie	Lampasas Cut Plains	Falls	366	19	19	43	19	0.00026
83050	Edwards Plateau	Lampasas Cut Plains	Coryell	609	12	9	52	28	0.00041
83051	Edwards Plateau	Lampasas Cut Plains	Coryell	609	12	9	52	28	0.00034
83052	Edwards Plateau	Lampasas Cut Plains	Lampasas	628	16	9	44	31	0.00030
83053	Edwards Plateau	Lampasas Cut Plains	Lampasas	628	16	9	44	31	0.00020
83054	Llano Uplift	Live Oak-Mesquite Savannah	Mcculloch	1278	9	4	41	46	0.00012
83062	Oak Woods	Mesquite Plains	Hood	343	29	19	37	14	0.00025
83064	Oak Woods	Mesquite Plains	Erath	355	21	14	47	18	0.00025
83065	Rolling Plains	Mesquite Plains	Palo Pinto	706	22	13	40	26	0.00029
83066	Edwards Plateau	Mesquite Plains	Callahan	636	22	11	42	25	0.00016
83067	Rolling Plains	Lampasas Cut Plains	Shackelford	2391	9	7	40	44	0.00014
83076	Rolling Plains	Mesquite Plains	Young	834	12	9	46	33	0.00010
83086	Edwards Plateau	Live Oak-Mesquite Savannah	Upton	6205	13	2	19	66	0.00008
83092	Rolling Plains	Live Oak-Mesquite Savannah	Garza	2190	8	6	30	56	0.00004
83112	Edwards Plateau	Live Oak-Mesquite Savannah	Crocket	12746	3	3	4	90	0.00005
83113	Edwards Plateau	Live Oak-Mesquite Savannah	Schleicher	2964	5	6	19	70	0.00008
83114	Edwards Plateau	Balcones Canyon Lands	Edwards	4232	8	2	21	69	0.00007
83139	Oak Woods	Lampasas Cut Plains	Williamson	298	28	17	38	17	0.00065
83140	Edwards Plateau	Balcones Canyon Lands	Hays	658	27	15	39	19	0.00067
83238	Edwards Plateau	Balcones Canyon Lands	Williamson	298	28	17	38	17	0.00110

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