QUANTIFYING FOREST VERTICAL STRUCTURE TO DETERMINE BIRD HABITAT QUALITY IN THE GREENBELT CORRIDOR, DENTON, TX

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This study presents the integration of light detection and range (LiDAR) and hyperspectral remote sensing to create a three-dimensional bird habitat map in the Greenbelt Corridor of the Elm Fork of the Trinity River. This map permits to examine the relationship between forest stand structure, landscape heterogeneity, and bird community composition. A biannual bird census was conducted at this site during the breeding seasons of 2009 and 2010. Census data combined with the three-dimensional map suggest that local breeding bird abundance, community structure, and spatial distribution patterns are highly influenced by vertical heterogeneity of vegetation surface. For local breeding birds, vertical heterogeneity of canopy surface within stands, connectivity to adjacent forest patches, largest forest patch index, and habitat (vegetation) types proved to be the most influential factors to determine bird community assemblages. Results also highlight the critical role of secondary forests to increase functional connectivity of forest patches. Overall, three-dimensional habitat descriptions derived from integrated LiDAR and hyperspectral data serve as a powerful bird conservation tool that shows how the distribution of bird species relates to forest composition and structure at various scales.

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CHAPTER 1

INTRODUCTION

Avian community structure is an established indicator of forest ecosystem health and overall biodiversity. Human activities have a great influence on bird community composition. For example, as a result of habitat destruction, degradation, or isolation, populations of many bird species have been declining for the past several decades, which can result in changes in community structure. In the past, many habitat studies have quantified and described habitat by intensive vegetative field sampling combined with spatial information derived from remote sensing. Quantifying habitat characteristics requires estimations of plant physiognomy (architecture) and community composition, both of which are the center of on-going ecological debates on the relative importance of each for determining bird habitat quality (Rotenberry, 1985; Fleishman et al., 2003; Walker, 2008; Muller, Stadler & Brandl, 2010). Uncertainties multiply when a set of estimations are introduced. For instance, the results of many habitat studies in forest ecosystems are often spatially coarse and lack accuracy in describing the canopy's vertical structure, which can be critical in determining bird habitat quality (MacArthur & MacArthur, 1961; Williamson, 1971; Anderson & Shugart, 1974; Rotenberry, 1985; Reid et al., 2004; Diaz, 2006).

This study examines how bird community structure associated with bottomland hardwood forests is related to local forest stand structure and spatial arrangements at larger areal extents, such as it occurs at landscape scales. Bottomland hardwood forest, which is characterized by seasonal inundation, is a valuable ecosystem that is rapidly disappearing and becoming increasingly fragmented in the central and southeastern US (Conner et al., 1998). Despite its numerous ecological functions such as flood mitigation, groundwater recharge, nutrient cycling, water storage, erosion control and wildlife habitat, the importance of bottomland hardwood forests has not been highly credited. It is estimated that most bottomland hardwood forests which once covered approximately 12 to 20 million ha have been reduced to 2.2 million ha by the late 1970s (Abernethy & Turner, 1987; Allen et al., 2004). Abernathy and Turner (1987) reported that the rate at which the bottomland hardwood forests disappeared between 1940 and 1980 was five times faster than other non-wetland hardwood forests. Bottomland hardwood forests have been steadily drained or converted to agricultural fields, residential/commercial developments, construction of flood defenses (Neal & Jemison, 1990; Allen, 1997; Conner et al., 1998; Allen et al., 2004). In Texas, nearly 63 % of the original extent of bottomland hardwood forests has been lost (Allen, 1997). In contrast to bottomland hardwoods in southeast US, which are threatened by agriculture, Texas-Oklahoma bottomland ecosystems are largely lost and degraded by reservoir constructions (Neal & Jemison, 1990).

Riparian forests are typically a small part of a landscape, yet provide rich habitats for many bird species due to their rich habitat components (Knopf, 1985). Their position in the landscape, overlapping with major water courses, also makes these forests act as migration pathways between stopovers. Moreover, despite the small sizes, riparian forests are good mediator to increase habitat connectivity to assist bird dispersal. Although the importance of bottomland hardwood forests or riparian forests is increasingly recognized (Twedt & Loesch, 1999; Bowen, Moorman & Kilgo, 2007), many habitat studies which employed remote sensing were limited. Lack of accuracy in describing vertical structure is due to the limited availability of remote sensing data, coarse spatial resolution of the data, and to the very nature of satellite imagery.

The site for this study is approximately 20 km^2 of the Greenbelt Corridor (GBC) of Ray Roberts Lake State Park, which surrounds the Elm Fork of the Trinity River, near Denton, Texas. Much of the bottomland hardwood forest surrounding the study site has been already been developed for residential, commercial, or flood control purposes, due to its close proximity to Dallas-Fort Worth metropolitan area. The remnant patches of bottomland hardwood forest serve as sanctuary for many local breeding species and migratory flyovers. With current human population growth, further loss, degradation, or fragmentation of remaining bottomland forests seems inevitable. Continuous isolation of forest communities leads to the demise of local populations and enhanced accelerating loss of biodiversity (Pimm et al., 1995). Hence, there is a need for habitat assessment to produce maps that not only display the known location of target species, but also the potential location of the species based on habitat properties at landscape level.

To achieve this objective, a bird census was conducted biannually during the breeding seasons of 2009 and 2010 in the GBC. A three-dimensional habitat map was then created by integrating two different kinds of remote sensing imagery. First, Light detection and range (LiDAR) data were used to describe vegetation height and then hyperion spectral data were used to delineate dominant vegetation types based on spectral resolution. The resultant LiDAR-hyperspectral integrated map, which has finer spatial and spectral resolution, was used in addition to vertical information to examine how bird community composition is related to forest stand structure and landscape heterogeneity. Integration of LiDAR with hyperspectral imagery can significantly increase our understanding of bird habitat niche requirements. Results provide useful new insights to the ongoing ecological debate on the relative importance of plant composition vs. physiognomy (architecture) of the vegetation to determine the birds' community assemblages (Rotenberry, 1985; Fleishman et al., 2003; Walker, 2008; Muller, Stadler & Brandl, 2010).

The importance of the habitat patch size and connectivity for forest interior birds in fragmented landscapes is well documented (Saunders, Hobbs, & Margules, 1991; Andren, 1994; Boulinear et al., 2001; Donovan & Flather, 2002; Balbontin & Ferrer, 2008). Larger tracts of mature forests tend to have more gaps and heterogeneous canopy vertical structure, both of which essentially create more favorable niche space for various species. On the other hand, highly fragmented forests without a deep core area or younger transitional forests with more homogeneous structure are known to create less favorable conditions for forest birds, resulting in lower species richness (Twedt & Loesch, 1999). At the local population level, habitat size is associated with the breeding success thus a probability of local extinction, while habitat connectivity is related to the dispersal success, which is reflected to the probability of colonization (Fahrig & Merriam, 1994; Saccheri et al., 1998; Opdam, 1998; Reed & Frankham, 2001).

Ultimately, effective management of remaining forests for avian conservation requires an investigation of functional heterogeneity of forest patches at landscape level. This investigation must consider the spatial arrangements of suitable habitats. Many landscape level studies which employed two-dimensional remote sensing imagery are limited by the lack of stand vertical information. Such studies can easily overlook the functional connectivity of forest patches. The degree of physical connectivity can be easily measured by distance between habitat patches, yet evaluation of functional connectivity is complex. Functional connectivity depends not only on the habitat composition or configurations, but also on the species specific response to these variables. For example, forest edge species or shrub species can benefit from increased area of edges caused by forest fragmentation and have lower extinction risk (McGarigal & McComb, 1995).

Previous studies have shown that human land use can place an irreversible impact on forest soils, plant community structure, and biodiversity (Dupouey et al., 2002). Further forest fragmentation seems inevitable around the study site due to close proximity to Dallas-Fort Worth metropolitan complex. However, at least within the study site, successive forest re-growth is occurring at once partially harvested sites. These secondary forests may buffer edge effects around fragmented mature forests and facilitate movements of sensitive species across dispersed habitat patches and may contribute to increase the local populations' capacity to persist in such environment. If habitat connectivity was enhanced by re-growth of secondary forests once harvested by humans, questions remain about when they start to function as effective corridors for target species. With that respect, three-dimensional habitat descriptions of the study site serve as a powerful bird conservation and management tool.

The goal of this study is to examine the relationship between forest stand structure, landscape heterogeneity, and bird community assemblages by creating multi-dimensional habitat maps using remote sensed imagery including LiDAR and hyperspectral data. It is assumed that fragmentation/degradation/isolation of mature bottomland hardwood forests is negatively affecting some local breeding birds as well as migratory flyovers who utilize the study site as a resting spot. A central challenge in assessing the effects of changes in habitat on bird community assemblages is to accurately describe bird habitat requirements at landscape level.

Specific objectives of this project are:

1. Integrate vegetation height information derived from LiDAR data and plant community classification derived from hyperspectral data, to produce a three-dimensional vegetation map.

2. Calculate a variety of forest patch metrics using the LiDAR-hyperspectral integrated map to quantify forest patch characteristics that may affect bird habitat quality.

3. Conduct a bird census biannually during breeding seasons and examine bird community structure (diversity, richness, and evenness) across different habitat types.

4. Assess the bird community responses measured by abundance, diversity, and evenness of the bird communities observed at each site, and then examine whether each of the community response variables is influenced by habitat types or/and sampling season.

5. Compare abundance, diversity, and evenness of communities of local breeding birds to that of non-local breeding birds (i.e., wintering birds or migratory flyovers).

6. Examine how the above measures of community structure (for both types of birds) are affected by habitat types and sampling season.

7. Examine the spatial distribution pattern of birds within forest stands.

8. Assess the spatial distribution pattern of local breeding birds across habitat types at landscape level.

9. Select a set of landscape characteristic (environmental) variables that best explains differences in bird community composition across habitat types.

10. Examine how much variation in bird community structure across habitat types is explained by environment alone, space, and spatially structured environmental variables.

11. Investigate how bird community composition, when grouped into foraging guilds, respond to selected habitat characteristic variables.

CHAPTER 2

LITERATURE REVIEW

2.1 Habitat Fragmentation

Human activities have greatly affected the preferred habitat of birds, often leading to declines in population sizes and changes in community structure. According to Audubon Society's nationwide bird population survey, the average population for common birds has declined 68 %, from 17.6 million to 5.35 million between 1967 and 2007 (National Audubon Society, 2007). Among many factors that contribute to the decline of bird populations, the major three factors that often act in concert are habitat loss, habitat isolation, and edge effects. The impact of habitat loss is direct and clear. It is not only the extent and quality of available habitat that influence bird population size and persistence, but also its distribution in the landscape. Habitat fragmentation, which is caused by the transformation of a landscape into smaller habitat patches, isolates patches from larger remaining tracts of intact habitat. As a result, habitat elements that are already patchily distributed in nature, such as wetlands, forests, and grasslands become smaller and further scattered in the landscape. The effects of habitat loss, degradation, and fragmentation on plants and animal populations have become an increasingly important concern (Meffe & Carroll, 1997; Fahrig et al., 1997; Riiters et al., 2002; Blair & Johnson, 2008). It is assumed that bottomland hardwood forests, including this study site, are particularly vulnerable to fragmentation, because they often occur as a relatively narrow strip along waterways. In the context of this study, the role of bottomland hardwood forest fragmentation is of particular concern in connection with the decline of some migratory bird species (Peterjohn & Sauer, 1994; Robinson et al., 1995; Sauer et al., 1999).

Although the population size of a particular species clearly depends on its life history requirements and competition, remnant habitat patch size influences the potential size of component species (Hanski, 1998; Robinson et al., 1995; Turner, Gardner, & O'Neill, 2001). Species confined within small patches, which often contain sub optimal habitat, face a greater risk of predation (Newton, 1998), higher exposure to adverse physical conditions (Andren, 1994), and higher risk of brood parasitism (Robinson et al., 1995; Blair & Johnson, 2008). All of these risks can lower adult survival probability and reproductive success, making the population more vulnerable to local extinction (Nicholas et al., 1985; Gill, 2006).

Population size and viability are influenced not only by the size or the shape of habitat patches, but the degree of habitat isolation that influences recolonization rates. The degree of habitat isolation is often measured by distances between the patches or the land use surrounding the habitat patch (Opdam, 1998; Boulinier et al., 2001). The idea of incorporating spatial configuration of habitat patches when assessing the quality of habitat is often used in metapopulation theory. This theory assumes some degree of gene flow among geographically separated populations that constitute networks (Hanski, 1998; Opdam, 1998). The theory treats a population as an ecologically and evolutionary functional unit and investigates the relationship between the target species' population dynamics at landscape level, rather than habitat patch level alone (Opdam, 1998; Saccheri et al., 1998; Reed & Frankham, 2001). In conjunction with source-sink dynamics that describe how variation in habitat quality may affect population dynamics, metapopulation theory has important implications for population persistence. Habitat quality varies among patches, thus it is important to consider the movement of organisms among the source (high quality habitat patch) and the sink (sub-optimum quality habitat patch).

Not only the removal of the source habitat but also the sink habitat influences dispersal and amplifies population fluctuations. Some species confined within networks of small isolated patches are less likely to receive migrants from other patches when the distance outweighs the dispersal ability of the species in question. Rapidly changing landscape structure can result in increased dispersal distance. If the rate of change in dispersal is not as fast as the rate of change in the landscape, isolated populations will not survive (Fahrig & Merriam, 1994; Morris, 1995; Saccheri et al., 1998; Hanski, 1998). The source-sink system can be also interpreted in the time domain: source patches act as persistent refuges, from which populations expand outward during good times, contracting back during less favorable times (Morris, 1995; Opdam, 1998). Further, spatial habitat fragmentation can aggravate temporal patchiness. For example, the effects of habitat fragmentation can be intensified when different phenologies generate mismatches between food supply and demand (Gill, 2006).

2.2 Floristic Composition vs. Forest Architecture

Since the classic work of MacArthur and MacArthur (1961) there has been an ongoing debate on whether bird community composition is determined by floristic composition of habitats or the physiognomy (physical structure) of habitats. MacArthur and MacArthur (1961) interpreted that the foliage height diversity contributed to the increase of potential bird niche space with an increase of physiognomic diversity of the vegetative community. Plant species' composition was considered the secondary determinant factor for the bird community assemblages (MacArthur & MacArthur, 1961; Williamson, 1971; Anderson & Shugart, 1974). Since these early studies, many researchers have examined the relative importance of plant species composition vs. physiognomy (architecture) of the vegetation to determine the diversity and composition of bird communities (Holmes & Robinson, 1981; Wiens & Rotenberry, 1981; Robinson & Holms, 1984; Rotenberry, 1985). Additional factors include foliage volume (Buchanan, Lewis, & Pierce, 1999), tree age (Sallabanks, Haufler, & Mehl, 2006), plant productivity (Cody, 1981), structure of shrub layer within stand (Reid et al., 2004; Diaz, 2006), plant succession including stand management (Sweeney et al., 2010), the size and configurations of the habitat patch, connectivity (Henderson, Merriam, & Wegner, 1985), and edge effects (McGarigal & McComb, 1995; Turner et al., 2001) and produced mixed results.

Three major factors contributed to vague results from the above cited studies. When assessing the functions of bird habitats, the ambiguous term “habitat” must be defined first. Ornithologists have been using the term "habitat" interchangeably to relate birds to certain aspects of the environment at various spatial and temporal scales depending on the context of the study. Spatially, the term "habitat" has been used to characterize foraging space (Robinson & Holmes, 1984), community (Pearman, 2002; Ritterhouse et al., 2010), and ecosystem (Johnston & Opdum, 1956; Ricklefs, 2001). Temporally, the environment is used seasonally (breeding, wintering, or migration), yearly, and historically (Wiens & Rotenberry, 1981; Newton, 1998). Second, the ongoing ecological debates on the relative importance of plant architecture (physiognomy) vs. plant floristic composition seem equivocal, because they are inherently positively correlated. For example, complexity in stand architecture increases as forest stands move through successional stages. Reciprocally, plant community increases in richness as plant succession continues.

Further, habitat isolation effects on bird populations in fragmented landscapes add complexity to the ongoing physiognomy vs. architecture debates which is already ambiguous by itself. For example, Henderson et al. (1985) argued that the degree of connectivity between forest patches could be more important than habitat characteristics to avian communities. As natural habitats become more fragmented by human activities, movements of individuals play an important role in maintaining local populations and ensuring genetic continuity. Genetic problems arise when movement of individuals making up a population is restricted among habitat patches that constitute a metapopulation and mating between close relatives becomes frequent. Populations that have undergone severe bottlenecks can be exposed to inbreeding depressions and reduce long term viability of populations (Saccheri et al., 1998; Reed & Frankham, 2001; Heber & Briskie, 2010). Dispersal or migration to maintain genetic continuity, however, is so costly that it set a spatial scale of habitat selection. Individual birds choose to disperse only when the increased fitness in the new home range compensates for the predation risks, time lost for reproduction, and uncertainties associated with the dispersal (Newton, 1998; Morris 2003). Clearly, habitat fragmentation increases the cost and risk associated with dispersal.

Fragmentation-induced habitat changes result in community structure changes. For example, alterations in landscape structure, such as urban patches limit the ability of some birds to disperse. Reciprocally, increase in urban patches or edge habitats around fragmented forests can favor some other species. This suggests that bird communities consisting of specific habitat requirements will have higher local extinction rates, while more adaptable and highly widespread tolerant species will benefit from habitat fragmentation. Habitat fragmentation or degradation thus can accelerate homogenization of birds' community assemblages, favoring generalist species over specialist species, changing bird community composition (Hinsley et al., 2002; Olden, 2006; Devictor et al., 2007; Devictor et al., 2008).

In addition, it is important to distinguish habitat and niche. According to Block and Brennan (1993), niche is defined as a set of "biotic and abiotic factors that permit an animal to use the environment." Niche also includes species' behavior to exploit the environment. Habitat and niche, therefore, are closely related, but not mutually exclusive concepts. Rather, habitat is essentially embedded within the niche. Suitable habitat includes those areas that have the physical conditions for a focal species to meet its requirements for survival and reproduction (Newton, 1998).

Habitat preference or selection, however, isn't directly translated into actual distribution of a species. Competition, for example, limits resource availability. Population dynamics combined with competition determine abundance and distribution of the individuals in a particular area (Hutto, 1985; Rosenzweig, 1981; Sergio & Newton, 2003). Behavioral responses to competition and available resources determine survival and fitness of the species (Block & Brennan, 1993; Jones, 2001). Factors that influence individual's habitat selection include experiences, innate morphological or physiological traits inherited from parents, or simple historical distribution consequences. Hutto (1985) classified these factors into intrinsic evolutionary constraints (i.e. how much food or protection is obtained within the habitat) and extrinsic constrains (i.e. habitat accessibility, travel time, predation risk, and weather patterns that may affect habitat use during migration). Individual species use selected habitats to meet life history needs (Block & Breenan, 1993). Thereby habitat selection entails complex processes acting at different organism levels and behavioral responses that are embedded within niche concept. Habitat is the end product of habitat selection (Jones, 2001).

2.3 Temporal and Spatial Scale Issues

Historically, ecologists have shown interest in the spatial distribution of organisms. In 1919, Clements claimed that climax was a predictable endpoint for plant communities, which was determined by the regional climate. In his theory, more focus was paid to the biotic processes driving succession, without stressing the importance of abiotic factors (Clements, 1919). This idea was countered by Gleason's individualistic hypothesis which argued that plant distribution was driven by individual responses to spatial gradients in the environment, reflecting spatial heterogeneity or gradients in resources (Gleason, 1939). Gleason's individualistic hypothesis gradually gained acceptance and was supported by published papers on various vegetation types (Egler, 1954; Whittaker, 1956). Since the replacement in ecological thought of Clements' climax vegetation with Gleason's individualistic hypothesis, focus of successional mechanisms have shifted to the contribution of the population dynamics of individual species (Peet & Christensen, 1980). An emerging concept of plants' spatial and temporal distribution was introduced by Watt in 1947, who explained the progression of successional stages as a pattern of forest patches across landscape. In other words, time and space were linked by Watt on a scale, which was defined as a landscape (Turner, 1989).

Driven by a need to understand ecological processes and changes in environmental on a larger scale, a growing number of studies in various disciplines have adopted a landscape ecology perspective (Turner, 1989; Schlosser & Kallemeyne, 2000; Farina, 2006). Landscape ecology emphasizes the interaction between spatial pattern and ecological processes. Reciprocally, it examines the cause and consequences of spatial heterogeneity across various scales (Turner et al., 2001).

The fundamental issue, attributed to the ambiguity of the ecological debate on habitat floristic composition vs. architecture, involves spatial scale. Scale is an essential topic in landscape ecology. It influences the conclusions drawn by an observer on whether the results can be extrapolated to other locations, extents, ecological systems, or times (Levin, 1992). Spatial analysis is increasing a focus in ecology, yet spatial analysis can take on a wide range of meanings, depending on the scale of the study and the context of discussion. Particularly, the appropriateness of spatial analysis substantially depends on the available data, which come in a variety of forms and scales. A central goal of ecology is to examine relationships between space and pattern. Because pattern is scale-dependent, the notion of pattern makes sense only at a particular scale. Evidently, the spatial extent of a landscape in which they are perceived varies among organisms and ecological processes.

Although the importance of spatial heterogeneity is widely recognized and employed in many ecological theories, such as succession, competition, and disease, the spatial variability of communities had not been studied until the 1980's. Many classic ecological studies assumed a uniform distribution of organisms within their geographic areas (Egler, 1954; Whittaker, 1956; MacArthur & MacArthur, 1961). Clearly, this assumption is hardly met for three reasons.

First, ecology is defined as the study of the interactions between biotic and abiotic elements and the abiotic environment is spatially structured, resulting in gradients or patchy distributions. Ecology also handles interactions of the various components of the ecosystem. The strength of interaction decreases as a function of distances between entities. Ecology often refers to processes that are explicitly spatial. Simple examples include seed dispersal, the spread of disease, and disturbance regimes such as fire. All of the three aspects of the spatial component of ecology can overlap and interact. Moreover, various agents act and generate patterns at different scales, or have competing effects on ecological patterns. For example, the spatial pattern of tree distribution may be elicited by the patchy distribution of resources such as soil nutrients, moisture levels, and light availability within a stand, leading to a clumped distribution pattern in areas with favorable environmental conditions. At the same time, competition for resources tends to maximize the distance between neighborhood individuals, creating a uniform distribution within resource patches. Furthermore, seed dispersal may interact with the pattern over multiple generations, possibly amplifying the magnitude of the patchiness or spatial heterogeneity.

For avian habitat studies, complexity of scale issues multiplies when bird habitat requirements vary spatially or temporally, depending on the resource availability, and the species of interest. Nevertheless, populations of the same bird species require similar environments and correspond with a spatially contiguous vegetation type required for survival and reproduction (Ricklefs, 2001). Therefore for the context of this study, habitat is defined as a space that comprises biological and physiological aspects contributing to the presence of species assembling a community. Assuming that habitat quality is deduced from bird abundance and community composition, accurate physical descriptions of habitat can increase our knowledge in bird habitat requirements.

2.4 Bottomland Hardwood Forests

Remnant patches of bottomland hardwood forests constitute part of the GBC. Bottomland hardwood forests are valuable ecosystems that are rapidly disappearing and fragmented in the central and southeastern US (Conner et al., 1998). The term bottomland hardwood forest is generally used to describe forest types that occur along rivers and streams on alluvial floodplains (King & Grant, 1996; Hodges, 1997). It also refers to a floodplain ecosystem dominated by trees with "morphological and physiological adaptations and/or reproductive strategies to perform certain life functions which enable the species to achieve maturity" under flooding conditions (cited in Conner et al., 1990). Bottomland hardwood forests are characterized by seasonal inundation by surface water and ground water during the growing season (Mitch & Gosselink, 2000).

It is estimated that the original extent of bottomland hardwood forests in the USA, which covered approximately 12 to 20 million ha, had been reduced to 2.2 million ha by the 1970's (Abernethy & Turner, 1987). Loss and alteration took a form of draining for agricultural purposes or conversion for residential and commercial developments (Turner, 1987; Allen, 1997; Conner et al., 1998; Allen et al., 2004; Neal & Jemison, 1990; Kellison & Young, 1997). Construction of flood control structures, surface mining, petroleum extraction, and urban development have accelerated the loss of bottomland hardwood ecosystem (Neal & Jemison, 1990). In Texas, nearly 63 % of the original extent of bottomland hardwood has been lost (cited in Allen, 1997). In contrast to bottomland hardwoods in the southeast US (which are threatened by agriculture), Texas-Oklahoma bottomland ecosystems are largely lost and degraded by reservoir constructions (Neal & Jemison, 1990). Bottomland hardwood forests provide important timber resources, water storage, nutrient cycling, water quality, erosion control, and wildlife habitat (Conner et al., 1990; Harris & Gosselink, 1990; Taylor et al.,1990; Kovacic et al., 2000; Allen et al., 2004).

Many researchers argued that hydrology, which include the flooding events, soil saturation, and the depth of water table, is the main force to change the community composition of the bottomland hardwood forest ecosystem (Mitsch & Gosselink, 2000; Kolka et al., 2000; Jones et al., 1989; Muzika, Gladden, & Haddock, 1987; Conner et al., 1990; Allen et al., 2004; Cronk & Fennessy, 2001; Simmons, Wu, & Whisesant, 2007). Hydrological conditions such as water depth, chemistry, and flow rates affect organic matter accumulation, primary productivity, microbiotic community, plant species composition, and successional direction (Messina &

Conner, 1998; Barnes et al., 1997, Cronk & Fennessy, 2001; Hupp & Bornette, 2003). Hydrology and plant community dynamics have reciprocal interactions, particularly at the water front. Riparian trees and shrubs play essential roles in enhancing flow resistance and sediment structure within the riparian zone and, thus, actively influencing rates of deposition and erosion (Francis, 2006; Corenblit et al., 2007). Additionally, at landscape level, evapotranspiration rates influenced by plants can cause changes in local water budget, possibly affecting hydrologic regime (Tabacchi et al., 2000; Corenblit et al., 2007). The temporal scale of this change extends from a single growing season, several years, to hundreds of years (Barnes et al., 1997; Hodges, 1997).

An important concept which needs to be considered is a temporal shift in species dominance or ecosystem change, summarized as plant succession (Smith & Houston, 1989). Plant succession involves changes in the community structure which are influenced by both abiotic and biotic factors, interacting simultaneously. These factors cause ecological succession through allogenic (external) and autogenic (internal) processes. Allogenic processes comprise hydrologic, climatic, and topographic changes, whereas autogenic processes include plant competition and species' life history characteristics (Cronk & Fennessy, 2001). Being an ecotone, or a transitional gradient between aquatic and terrestrial environments (Mitsch & Cosselink, 2000), a bottomland hardwood ecosystem interacts with allogenic forces at both ends of the ecotone. The community composition tends to be dominated only by water tolerant species when water level rises, while it approaches toward an upland forest during droughts. Succession in southern riparian forests is highly influenced by the degree of drainage (Hodges, 1997). According to Hodges, on a poorly drained site, species composition changes from Black willow (*Salix nigra*)-Box elder-Swamp privet-Buttonbush mixed stands, Elm-Ash stands, to Oak-Elm association. On a well-drained site, Eastern cottonwood is replaced by Box elder, Sycamore-Pecan-Elm mixed stands, Elm-Ash stands, to Oak-Hickory-Elm association. Hodges (1997) also pointed out that the Elm-Ash associations that exist in both successional patterns are capable of self-replacement. Although it is not a climax forest, it will persist as long as the condition allows.

Barry (2000) conducted a systematic bird census at the GBC on a line transect along the Trinity River Trail and recorded 21 bird species during the 1999 and 2000 breeding seasons. Of the bird species observed, forest interior breeding birds and neo-tropical migratory birds are experiencing a nationwide decline in population size (Robinson et al., 1995; Sauer et al., 1999). Barry investigated the effect of habitat heterogeneity across the landscape on the birds' occurrences using Landsat Thematic Mapper (TM) images with a spatial resolution of 30 m. His study suggested that the birds within patch forests are somewhat insulated from horizontal landscape-scale effects, but the relationship between the riparian forest spatial configurations and bird species diversity remained unclear. Barry’s study was limited by the lack of information regarding stand structure and the coarse spatial resolution of the remote sensed images available at that time.

The study site, the GBC, consists of a diversified ecosystem that includes remnant patches of riparian forests. Although riparian forests are typically a small part of the landscape, they provide rich habitats for many bird species due to their rich habitat components (Knopf, 1985). Their position in landscape, overlapped with major water courses, also makes the riparian forests act as migration pathways between stopovers. Several species of birds either inhabiting or utilizing riparian forests as migration pathways have undergone population declines. Of the species inhabiting in the GBC area, field sparrow (*Spizella pusilla*), eastern meadowlark (*Sturnella magna*), dickcissel (*Spiza americana*), red-headed woodpecker (*Melanerpes* *erythrocephalus)* (National Audubon Society, 2007), prothonotary warbler (*Protonotaria citrea*) (Sauer et al. 1999), wood thrush (*Hylocichla mustelina*) (Peterjohn & Sauer, 1994) are experiencing a decreasing trend. In addition, some local breeding species such as northern cardinal (*Cardinalis cardinalis*) (Robinson et al., 1995), indigo bunting (*Passerina cyanea*) (Payne & Payne, 1998) and red-eyed vireo (*Vireo olivaceus*) are highly vulnerable to habitat fragmentation. Habitat fragmentation and associated reduction in food supply, foraging space, and nesting space, can result in population declines which can ultimately reduce fecundity of the species (Robinson et al., 1995).

It is important to note that population trends vary among species. Some generalist species that are more adaptable to landscape changes are known to benefit from increased edges (Hinsley et al., 2002; Devictor et al., 2007; Devictor et al., 2008; McWethy et al., 2009). Even neotropical migratory species, red-eyed vireo, which often co-occur with wood thrushes, has increased 1.4% annually during the same period while wood thrushes decreased 1.7% (Sauer & Peterjohn, 1994). This illustrates that population persistence is influenced by many factors, thus quantifying habitat quality deduced from birds' occupancy is challenging. Further, each species has uniquely different habitat requirements, habitat ranges, and spatial and temporal migration patterns. Thus migratory bird populations could be limited by events on the North America breeding grounds, events during migration, or events on wintering grounds, all of which could influence their survival and reproduction success. Regardless, it is assumed that fragmentation, isolation, and degradation of North America breeding grounds or migration stopover sites have adverse effects on many bird species. A central challenge is first to accurately capture the bird habitat requirements and then assess mechanisms causing any habitat deficiencies.

2.5 Avian Habitat Assessments and Remote Sensing

Quantifying habitat quality meaningful to the species of interest is difficult. For birds, this difficulty coincides with the complexity of habitat requirements, which is influenced by both the vertical and horizontal structure of the vegetation communities. LiDAR is an emerging remote sensing technology. It is an airborne scanning laser system that is capable of describing horizontal and vertical information at high spatial resolution and vertical accuracies (Lefsky et al., 2002; Kato et al., 2009). LiDAR remote sensing has been increasingly used in many forestry applications describing forest vertical structures such as tree height, stand surface structure, stand density, biomass, basal area, and architectural structure consisting of multiple vegetation layers within a stand (Leckie et al., 2003). LiDAR remote sensing uses a laser light to detect distance by measuring the travel time of the light (Jensen, 2007). LiDAR remote sensing has also started to be accepted for use in wildlife habitat studies to connect the relationship between the target species and its habitat requirements (Lefsky et al., 2002; Hinsley et al., 2002; Baltsavias et al., 2006; Bergen, Gilboy, & Brown, 2007). Particularly, researchers have been successful in establishing a correlation between vertical vegetation structure and bird species occurrence (Hinsley et al., 2002; Bergen et al., 2007; Goetz et al., 2007; Clawges et al., 2008; Smart et al., 2012).

Remote sensing of vegetation focuses on the absorption and scattering properties of target surfaces. The absorption and scattering properties of surfaces are defined by their chemical properties and three dimensional structures such as leaf shape or canopy geometry, which determine the reflectance spectra. Analysis of these spectra enables mapping of biogeochemical features of plants (Ustin et al., 2004; Jensen, 2007). Traditionally, monitoring vegetation health at landscape level has been achieved by using multispectral remote sensed images that are generally composed of 5 to 10 bands of relatively wide bandwidth (70-400 nm). Many vegetation studies have been conducted to measure relative abundance and activity of green vegetation based on the chlorophyll absorption approximately at the spectral wavelength of 0.68 um (Ustin et al., 2004; Jensen, 2007). Detecting spectral reflectance from healthy vegetation and chlorophyll absorption in vegetation multispectral remote sensing have been used for generating broadly classified forest cover type maps. Using landscape metrics derived from satellite imagery and aerial photography, researchers have detected the spatial extent of forests and changes over time (Wulder et al., 2004; Jensen, 2005). However, results are not robust in providing more detailed or species-level information. The resolution of the products, both in spatial and spectral, has proven to be the major limitation of these studies. For a feature to be detected, its size generally has to be equal to or larger than the resolution cell. If the feature is smaller than this, it may not be detectable as the average brightness of all features in that spatial resolution cell will be recorded (Wulder et al., 2004). Similarly, average spectral information generated based on broad bandwidths, often result in loss of the critical information that might have been captured in specific narrow bands (Dalponte et al., 2009).

Hyperspectral remote sensing is a technology, established in the 1990s, that measures and processes reflected sunlight in hundreds of narrow and continuous spectral bandwidths (4-10 nm). These narrow spectral bandwidths can be used to identify specific and subtle vegetation constituents. Successful applications include pigment composition and content, canopy water content, and foliar chemistry (Blackburn, 1998; Ustin et al., 2004; Asner & Martin, 2008; Martin et al., 1998), all of which are crucial for providing additional information in quantifying biophysical characteristics of vegetation and differentiate vegetation types which are similar in nature. Hyperspectral data has been used to determine species level abundance patterns (Martin et al., 1998; Clark et al., 2005; Plourde et al., 2007; Dalponte, 2008) and forest structure (Ustin & Trabucco, 2000). Accurate spectral response derived from hyperspectral data that comes in hundreds of narrow contiguous spectral bands allows users to detect differences in the tree species, yet is inseparable from data redundancy. Reducing the dimensionality of the data to separate the most meaningful spectral response is a common approach to working with hyperspectral datasets (Williams & Hunt, 2002, Anderson et al., 2008).

LiDAR and hyperspectral data contain different and complementary information: LiDAR provides detailed information about the vegetation height but no information on the spectral reflectance, whereas hyperspectral imagery offers a detailed description of the spectral signatures of forest covers but no information on the height of the plants. It is, therefore, hypothesized that a three-dimensional map incorporating LiDAR-derived vegetation height will provide more realistic representation of how birds perceive, utilize, and interact with landscape patterns. Information on vegetation height can be used to calculate vegetation volume and foliage height diversity.

Although the importance of vertical perception when quantifying the bird habitat has long been recognized, incorporating three-dimensional information to habitat studies using remote sensing technologies have been limited by data availability. In recent years, increasing numbers of bird habitat studies have combined LiDAR with spectral information on vegetation. For example, Hyde et al. (2006) demonstrated that fusion of LiDAR canopy height and spectral information derived from Landsat Enhanced Thematic Mapper (Landsat ETM) significantly increased the tree classification accuracy for wildlife management in a fir-dominated forest in the Sierra Nevada Mountains of California. Similarly, Goetz et al. (2007) integrated LiDAR with Landsat ETM to investigate bird species richness and diversity in eastern temperate forests in

Maryland. Further, Clawges et al. (2008) investigated the bird habitat requirements using LiDAR and multispectral remote sensing imagery (IKONOS) in a pine-aspen forest in South Dakota. All of these studies have concluded that vertical forest structural information combined with vegetation spectral information using remote sensing technology can improve the accuracy in describing wildlife habitat requirements. However, integration of LiDAR and spectral information from other sensors is yet to be demonstrated for bottomland hardwood forests. Previous studies have shown that bottomland hardwood forest, a rapidly diminishing ecosystem in the US, serves important breeding grounds for various bird species including neo-tropical songbirds, and other migratory stopovers. It is anticipated that integration of LiDAR with other remote sensed images that have finer spatial and spectral resolution data allows for detailed physiological information on vegetation, hence significantly increase our understanding of bird habitat niche requirements in bottomland hardwood forests.

CHAPTER 3

MATERIALS AND METHODS

3.1 Study Site

The study site, approximately 20 km^2 of the Greenbelt Corridor (GBC) of the Elm Fork of the Trinity River, lies between the upper side of Lewisville Lake at US Highway 380 and the Lake Ray Roberts Dam at Farm Road 428. The area is owned by the US Army Corps of Engineers. The eastern side of the GBC is managed by Texas Parks and Wildlife Department (TPWD) and the western side is managed by City of Denton. Until 1980s when it became a part of Texas State Park System, this area was partially used for agriculture and pastures. Historical aerial photos show that the GBC also contains mature bottomland hardwood forests where no agriculture is known to have occurred. Former agricultural sites have been progressively transformed back to bottomland hardwood forests, resulting in forest patches in distinctive age cohorts ranging from decades to over hundred years.

Dominant tree species of bottomland hardwood forests are cedar elm (*Ulmus crassifolia*), hackberry (*Celtis reticulata*), and green ash (*Fraxinus pennsylvanica*) with an occasional occurrence of bur oak (*Quercus macrocarpa*), pecan (*Carya illinoensis*), and eastern cottonwood (*Populus delitoides*). At maturity, these species in bottomland hardwood forests can easily exceed 20 to 30 m in height. Below these overstory species, black walnut (*Juglans nigra*), osage orange (*Maclura pomifera*), box elder (*Acer nigra*), and hawthorn (*Crataegus texacana*) comprise the understory. The upland forest is dominated by post oak (*Quercus stellata*) and hackberry (*Celtis reticulata*). Throughout the study site, the herbaceous layer underneath trees consisted mainly of common greenbrier (*Smilax rotundifolia*), poison ivy (*Rhus toxicodendron*), and Virginia wild rye (*Elymus virginicus*) (Barry & Kroll, 1999).

The major natural disturbance for this site is a series of flooding events. The site is used as flash-flood mitigation by the US Army Corps of Engineers. In the event of seasonal heavy and long rainfall, Clear Creek flow combined with Ray Roberts Dam release submerges large portions of the GBC for at least a few months. The latest flooding event occurred in summer 2007. Ecologically, the GBC is associated with the Eastern Cross Timber ecosystem (Bailey, 1995), much of which has been developed for residential and commercial purposes due to its close proximity to Dallas and Fort Worth. The GBC has remnant patches of bottomland hardwood and upland forests, which consists of a forest mosaic comprising different vegetative age cohorts including wetland and grassland habitat (*Figure 3.1*). Even though the site is moderately fragmented, it still contains a diversified ecosystem that makes the GBC an important sanctuary for both non-migratory and migratory birds.

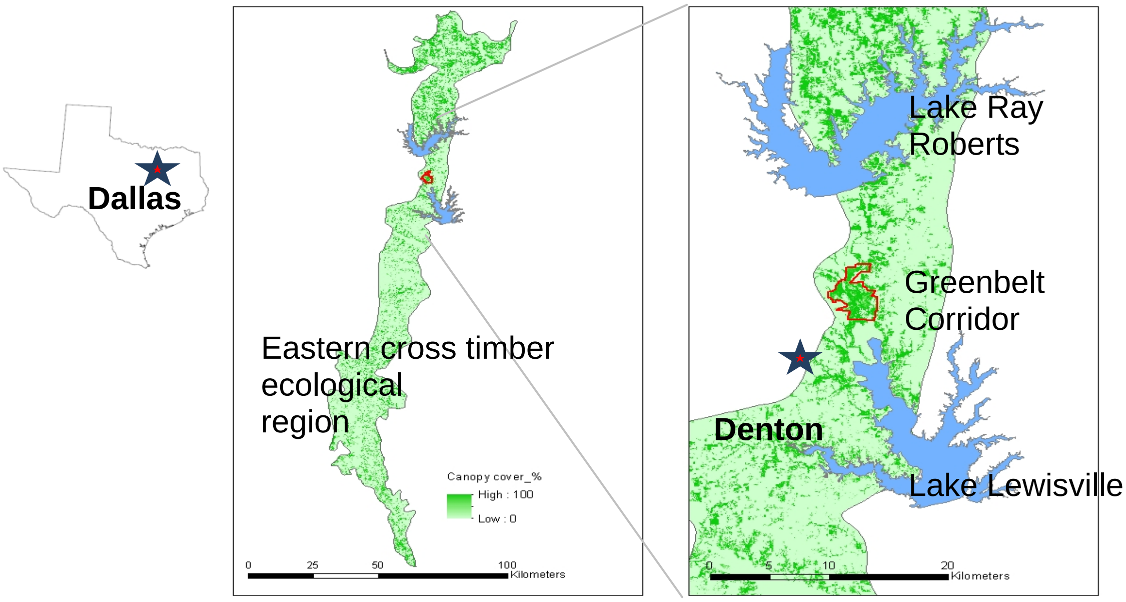


Figure 3.1. Forest canopies remaining in Eastern Cross Timber Ecoregion (left) and in surrounding areas of the GBC (right).

3.2 Habitat Mapping

3.2.1 Remote Sensing Data

3.2.1.1 LiDAR Data

Tree height and stand structure is essential to describe vegetative composition associated with forest bird niche requirements. To quantify vertical structure of forest stands, LiDAR data were acquired from Texas Natural Resources Information System (TNRIS). This LiDAR point cloud was collected on January 29, January 30, and March 3, 2011 using a kinematic global positioning system (GPS) on an aircraft. Point cloud density ranges from four to six points per square meter. As the aircraft moves forward, a scanning mirror directs pulses of laser light and this reflection information is processed to determine the height of the target hit by the laser. Theoretically, the first object hit by the laser is the outer surface of trees and the last object hit by the laser is the ground surface. Raw data for some distinctive site characteristics such as vegetation surface, water, and ground were displayed in *Figure 3.2*. Computer software, MARS 7.0 was used to visualize raw LiDAR point clouds.

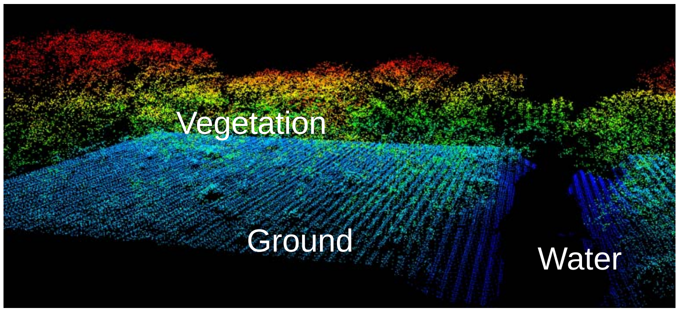


Figure 3.2. Surface structures in the GBC. The water areas correspond to Clear Creek.

The raw data were processed using ESRI ArcGIS10.0. Inverse distance weighting (IDW) spatial interpolation method was used to create a 1×1 m raster out of LiDAR point clouds. After converting point clouds to raster format, the actual height of tree, or canopy height model (CHM) was calculated by subtracting the digital elevation model (DEM), or ground surface from the digital surface model (DSM) by the equation CHM = DSM-DEM, where canopy height model (CHM) is the actual height of tree, the digital surface model (DSM) is the surface height, and digital elevation model (DEM) is the ground elevation (Jensen, 2007).

3.2.1.2 Hyperspectral Image

Hyperion is a satellite hyperspectral sensor at a 705 km orbit covering the 0.4 to 2.5 µm spectral region with approximately 242 spectral bands at 10-nm spectral resolution (Jensen, 2007). This spectral region, much wider than the human visible light region, provides detailed information about the target object, as compared to multispectral images such as the digital photos or conventional Landsat TM images composed of seven bands. Although the potential of hyperspectral image in land cover classification is widely accepted, application is limited due to the enormous volume of the data that complicates image display, classification, and analysis (Martin et al., 1998; Plaza et al., 2004; Jensen, 2005). Large numbers of bands can provide detailed spectral information on the ground cover classification, yet can create redundancy (Jia & Richards, 1999). Choosing sufficient and correct training samples to meet the classifiers' objective is essential. If too many or redundant bands are selected in classification, results can be hard to interpret.

Hyperspectral imagery used for this study was acquired on August 30, from NASA's EO-1 with latitude of 33'20" North and longitude of 97'10" West. The image has 242 bands and contained a spatial resolution of 30 m. Computer software ERDAS IMAGINE 2011 was used to process, visualize, and analyze hyperspectral imagery. Atmospheric correction and geometric correction reduced the band numbers from 242 to 198. Principal component analysis (PCA) was then employed to examine more relevant band combination to delineate vegetation. PCA reduces the original numbers of the variables to a smaller set of new, orthogonal (uncorrelated) components, or eigenvectors of the covariance matrix, that will account for a large part of the total variation in the original dataset (Carr, 1995; Bell & Baranoski; 2004). After applying PCA, a supervised classification method with maximum likelihood algorithm will be applied to map vegetation.

3.2.2 Integrating LiDAR and Hyperspectral Imagery

LiDAR and hyperspectral data were integrated by a simple procedure. First the dimensionality of hyperspectral data was reduced by PCA. LiDAR vegetation height information was then added to the five hyperspectral PCs. Integration of two images was performed by ESRI ArcGIS10.0.

3.2.3 Calculation of Forest Patch Metrics

The integrated LiDAR-hyperspectral imagery served as a base layer to measure configurations of forest patches contained within 100 m radius circles surrounding bird census points. Statistical software, Fragstats 4.1 (McGarigal, Cushman, & Ene, 2012) was used to calculate a variety of forest patch metrics for three tree height classes: 10 m, 15 m, and 20 m. Calculated forest patch metrics are summarized in Table 3.1.

To visualize output, forest patch metrics were organized by using Microsoft Office Access and then transformed to ESRI ArcGIS10.0.

Table 3.1

Forest patch metrics used to describe habitat quality.

	Description of the variables
Habitat composition	
% landscape, 10m	% area covered by trees > 10m in height
% landscape, 15m	% area covered by trees > 15m in height
% landscape, 20m	% area covered by trees > 20m in height
Habitat types (6 types)	Water, Grass, Wetland forest, Sec BHF, Mature BHF, Oak-elm
Habitat configurations	
Largest patch index, 10m	% of total area comprised by the largest patch > 10m in height
Largest patch index, 15m	% of total area comprised by the largest patch > 15m in height
Largest patch index, 20m	% of total area comprised by the largest patch > 20m in height
Clumpiness, 10m	Frequencies with patch > 10m in height apprear side-by side
Clumpiness, 15m	Frequencies with patch > 15m in height apprear side-by side
Clumpiness, 20m	Frequencies with patch > 20m in height apprear side-by side
Connectance, 10m	% of finding forest patches > 10m in height within 10 m distance
Connectance, 15m	% of finding forest patches > 10m in height within 15 m distance
Connectance, 20m	% of finding forest patches > 10m in height within 20 m distance
Nearest neighbor distance, 10m	Distance to the nearest neighbor forest patches > 10m in height
Nearest neighbor distance, 15m	Distance to the nearest neighbor forest patches > 15m in height
Nearest neighbor distance, 20m	Distance to the nearest neighbor forest patches > 20m in height
Mean height	Mean vegetation height
Maximum height	Maximum vegetation height
Coefficient of variation	Coefficient of variation of tree height

3.2.4 Forest Area Change Detection

Changes in land use between 1984 and 2006 around the study site was assessed by comparing 1984 Landsat Thematic Mapper (TM) imagery and 2006 National Land Cover Data set (NLCD). The 1984 image was first registered and then classified into six land cover types: bare, water, development, forest, shrub/herbs, and pasture by using supervised classification. The 2006 NLCD was reclassified to six land cover types accordingly. Changes from one land cover type to the other were detected by overlaying the two images.

3.2.5 Accuracy Assessment

3.2.5.1 LiDAR Height

An accuracy assessment on canopy height derived from LiDAR was conducted at seven forested sites out of 42 bird census points in summer 2011. At each site, all trees within a 25×25 m quadrat plot established around the center of bird census plot were identified and its height and diameter at breast height (DBH) were measured. Tree height was measured using a clinometer or a range finder. Location of trees whose DBH was greater than five cm was manually digitized using ArcGIS 10.0. The relationship between LiDAR- derived vegetation height and ground reference information was summarized to assess LiDAR- derived vegetation height accuracy.

3.2.5.2 LiDAR-Hyperspectral Integrated Map Classification

Another accuracy assessment was conducted to examine classification accuracy of the LiDAR-hyperspectral integrated imagery. As discussed in the previous section, the LiDAR-hyperspectral integrated imagery was classified to six bird habitat types: water, grassland-herbaceous community, wetland forest, consisting mainly of swamp privet, secondary bottomland hardwood forest, mature bottomland hardwood forest, and oak-elm association. A total of 200 random points were generated across the study site. An error matrix was produced to assess the relationship between the 200 random points generated on the classified map and the field knowledge. Kappa coefficient ($\hat{\kappa}$) was computed by the equation;

$$(\hat{\kappa}) = \frac{N\sum_{i=1}^{k} x_{ii} - \sum_{i=1}^{k}(x_{i+} \times x_{+i})}{N^{2} - \sum_{i=1}^{k}(x_{i+} \times x_{+i})}$$

where k is the number of rows (habitat classes) in the matrix , x_{ii} is the number of observation in row i and column i, and x_{i+} and x_{+i} are the marginal totals for row i and column i, respectively, and N is the total number of observations (Cognalton, 1991; Jensen, 2005).

Kappa statistics ($\hat{\kappa}$) values greater than 0.8 (80%) represent strong correlation between the classified map and the ground reference information. Whereas, kappa statistics ($\hat{\kappa}$) values greater than 0.4 (40%) represent moderate agreement between the map and ground reference (Jensen, 2005).

3.3 Bird Data

3.3.1 Bird Census

Using a pre-classified vegetation map based on Landsat TM images, the study site was classified to five bird habitat types or physiognomies: mature forest, young forest in transitional stage, wetland forest, grass-herb, and water. The assumption behind this classification is that bird habitat quality is influenced by vegetation height across physiognomies as well as the horizontal connectivity of each other. It is anticipated that mature forests with multiple vegetation layers of overstory, understory, shrub, and forbs have more heterogeneous stand structures, thus allowing for greater numbers of species. Dense, young, and structurally monotonous patches, on the other hand, may serve as temporary shelters, but will be less favorable habitats.

Following the United States Forest Service (USFS) bird census protocol (Ralph, Sauer, & Droege, 1995), a total of 42 bird observation stations were randomly selected using a stratified random sampling design within the study site in an attempt to standardize the species richness across varying vegetation types. GPS coordinates of each site is provided in the Appendix B. The number of bird observation stations thus reflects the percentage of potential bird habitat types in area. The location of bird census points and the study site is shown in Figure 3.3. Each sampling station is at least 250 m apart to minimize the possibility of double-counting the same individuals. In addition, each station is at least 500 m apart from the major road with high traffic to minimize the noise bias during point count surveys. Furthermore, each station is at least 20 m away from the edge of the pre-classified vegetation types in order to avoid the potential misclassification of edge vegetation types, which is a common issue when using Landsat TM images.

At each observation point, observers face to the north and record the bird species, numbers, and approximate location of individuals for 10 minutes. 25 m, 50 m, and 100 m radial circles in order were used to estimate the distance to the center of the plot to the individual bird detected. Position of singing male, simultaneous song of two males, known change in position, and nest site along with time and weather conditions were recorded (See Appendix A). The bird survey was conducted from the dawn to 10:30 AM only under favorable weather conditions. The survey was canceled if unfavorable weather conditions arose because bad weather can reduce the accuracy of species identification as well as affect bird behaviors. All stations were visited three times for one year in order to standardize the avian species composition that may change over time. Each visit represented winter to early breeding, migration, and peak breeding season, respectively. Unknown species calls were recorded using the Marantz PMD 660 digital recorder and the Audio-Technica AT815B - Line/Gradient Shotgun Condenser Microphone and carefully identified in the lab with help from experienced local ornithologists. Detected bird location and species were summarized in a database, and digitized using ArcGIS programs to display on the map.

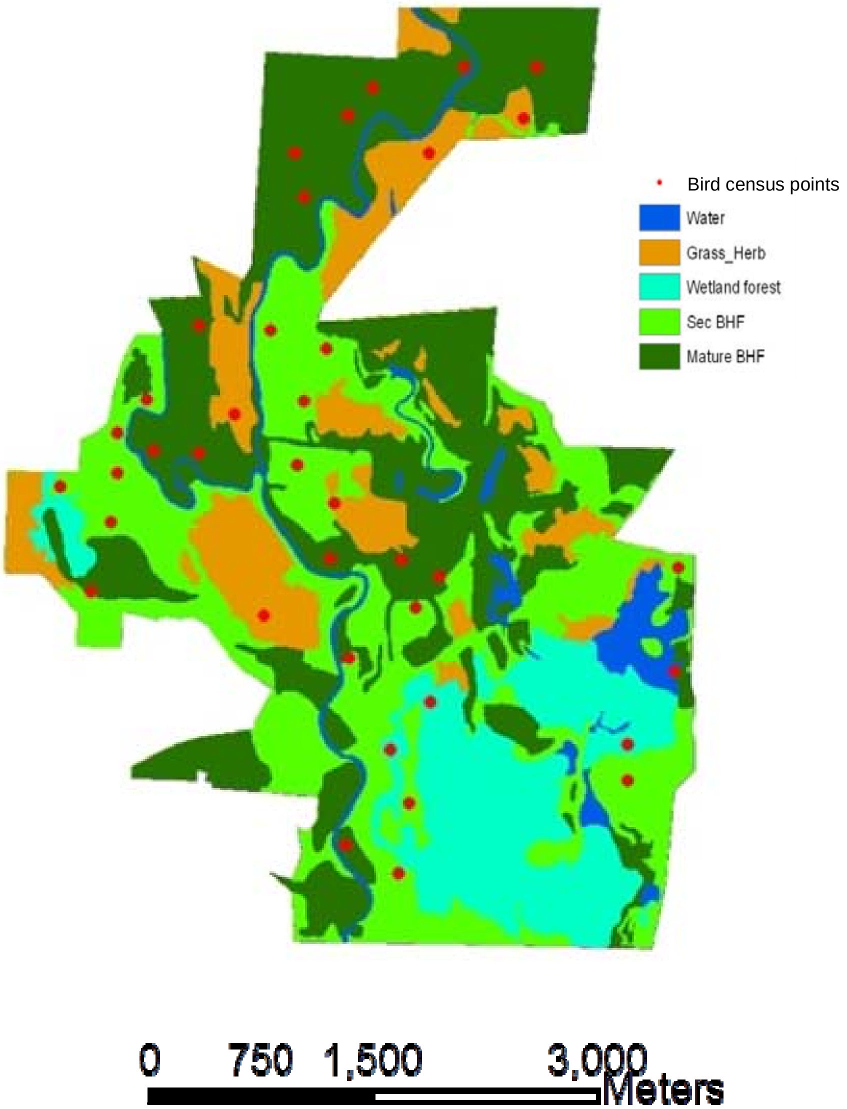


Figure 3.3. Location of bird census points across five habitat types. Classification was performed based on an existing two-dimensional map.

3.3.2 Bird Census Data Analysis

3.3.2.1 All Birds

A variety of community parameters across habitat types was calculated using bird census results: abundance (the number of individuals present), species diversity, richness, and evenness. For species diversity indices, which are widely used by ecologists to measure the degree of contribution of each species, weighted by its relative abundance, Simpson's index and the Shannon-Wiener index were used. Simpson's index (D) is calculated by the equation

$$D = \frac{1}{\sum p_i^2}$$

where p_i is the proportion of each species in the total samples of individuals. For any particular number of species in a sample (S), the value of D varies from 1 to S, depending in species abundances. When all species have unequal abundances, D becomes less than the number of species. Rare species contributes less to the value of the diversity index than common species (Babour et al., 1999).

The Shannon-Wiener index (H) is calculated by the equation

$$H = -\sum p_i \ln p_i$$

where pi = the proportions of each species in the total samples of individuals. Higher values of H indicate greater diversity. As is the case of Simpson's index, the Shannon-Wiener index gives less weight to rare species because H is approximately proportional to the logarithm of number of species.

Both diversity indices are based on the proportions of each species in the total samples of individuals, thus a function of numbers of species present. It is mathematically inevitable that the number of species in a sample increases with the number of individuals sampled (Krebs, 1998). Species diversity increases with sample size, and differences in diversity could be caused by differences in sample size.

Species evenness (equitability) was calculated using Pielou's evenness (J) by the equation;

$$J = \frac{H}{\log(S)}$$

where H is Shannon Wiener index and S is the total number of individual species. Pielou's evenness (J) is constrained between 0 and 1. It would be always zero when S=0 or 1, and will reach 1 for maximum diversity.

3.3.2.2 Local Breeding Birds vs. Others

There is a great variation in bird habitat range and migration distances among families, populations, and individuals. Detected birds in the study site include year-round residents, leap-frog migrants, and long distance migratory stopovers. To examine whether certain habitat types accommodate more bird species, all birds were classified into two groups; the first group consists of local breeding birds including year-round local residents and neo-tropical songbirds that migrate from the south to breed in the study site. The other group contained both wintering species and migratory flyover that don't necessarily breed in the study site, but used the site to forage or to rest. Classification was carried out based on the bird habitat range map prepared by Nature Reserve and provided by the Cornell Lab of Ornithology (Cornell Lab of Ornithology, 2013). After classifying birds into two groups, species richness, diversity and evenness indices were calculated to assess differences in birds' responses to habitat types and sampling seasons. Species diversity was assessed by Simpson index (D) and Shannon-Wiener index (H), evenness was calculated by using Pielou's evenness (J). A community ecology package, Vegan program (version 2.0-7) package developed for a statistical software R (version 2.15.2) (Oksanen, 2013) was used for calculation and statistical analysis.

3.3.3 Bird Community Composition across Habitat Types

To visualize differences in local breeding bird community structure across habitat types, non-metric multidimensional scaling (NMDS) function was used. Dissimilarity across habitat types was first measured using Bray-Curtis dissimilarity index across habitat types and then results were plotted in the ordination space. Bray-Curtis distance (d_{ij}) was calculated by an equation:

$$d_{i,j} = \frac{\sum_{k=1}^{n} \left| y_{i,k} - y_{j,k} \right|}{\sum_{k=1}^{n} \left(y_{i,k} + y_{j,k} \right)}$$

that measures dissimilarity between the observations *i* and *j*. Here *k* is the index of a variable and *n* is the total number of variables *Y*. After regressing dissimilarities in species assemblages on habitat types, NMDS calculates a new distance δ_{ij} between the observations in the initial

$$\hat{\delta}_{ij} = \hat{\beta}_o + \hat{\beta}_1 d_{ij}$$

configurations and again regress on d_{ij}. The result of this regression is a set of values $\hat{\delta}_{ij}$ given by

Further, NMDS computes a goodness of fit between δ_{ij} and $\hat{\delta}_{ij}$ as a stress:

$$\text{Stress} = \sqrt{\frac{\sum_{i=1}^{m} \sum_{j=1}^{n} \left(\delta_{ij} - \hat{\delta}_{ij} \right)^2}{\sum_{i=1}^{m} \sum_{j=1}^{n} \hat{\delta}_{ij}^{2}}}$$

The amount of stress can be used for judging the goodness of fit of NMDS. A small stress value indicates a good fit, whereas a high value indicates a weak fit. Kruskal (1964) provided for interpretation of the stress value with respect to the goodness of fit of NMDS. According to

Kruskal, a stress level higher than 0.2 is considered as weak fit, 0.1 is fair, 0.05 is good. Although the amount of stress is informative, it has been generally accepted that stress level only offers a vague indication of goodness of fit (Oksanen, 2013). To further test whether there is a significant difference in species assemblages among groups, analysis of similarity (ANOSIM) was employed. ANOSIM was calculated by an equation:

$$R = \frac{\widehat{r_B} - \widehat{r_W}}{\frac{n(n-1)}{4}}$$

where *n* is the sample size and $\widehat{r_B}$ and $\widehat{r_W}$ are the mean of the ranked similarities between groups and within groups, respectively. *R* scales from -1 to +1, with 0 indicating completely random grouping. Statistical significance was calculated based on 1000 Monte Carlo permutations (Oksanen, 2013).

NMDS function in the R-vegan package also produces plots that can be used for visual interpretation of differences in bird assemblages across habitat types. Because NMDS geometrically arranges sites so that distances among sites in the graph reflect their ecological distances. Sites that are close together in the ordination space are interpreted as being similar in bird species composition, whereas separated sites are interpreted as consisting of different bird species composition.

3.3.4 Spatial Distribution Pattern of Birds

3.3.4.1 Point Pattern Analysis in Ecology

In ecology, point pattern analysis is commonly applied to map plants distribution (Perry, Miller, & Enright, 2006). The aim is to infer the mechanisms that generate the distribution pattern. In plant communities, above-ground and below-ground competition for resources is presumed to lead to uniform distributions when plants exclude competitors from their immediate neighborhoods. At the same time, resources such as nutrition, water, or light availability might be patchily distributed, which would result in contiguous distribution of plants corresponded to patchy resources. Additionally, seed dispersal would cluster plant communities as well. Thus, plant communities tend to be spatially patterned in different ways, as affected by different agents, so the task is to deduce what agents, at which scales, are contributing to the pattern formation (Barbour et al., 1999).

In animal ecology, point pattern analysis is less common, yet there are some studies on territorial species. In territorial species such as songbirds, point pattern analysis has been used to describe the over-dispersion to show territorial defense (Melles et al., 2009). In species that flock or congregate in resource patches, point pattern analysis could also be used to quantify the clustering pattern.

Point pattern analysis comes in variety of forms, but most share a few basic features. The simplest method is to relate point density to area per point. Area per point provides the expectation about distances between points, often used as an expected nearest neighbor distance (NND). Nearest neighbor statistics compared the observed NND for a set of points to the NNDs expected for completely random points. This way, it can be tested whether points of target are closer together or farther apart than expected by chance. NND statistics only measure distance to the nearest neighbor thus cannot indicate the scales of pattern. This limitation motivated the development of Ripley's *K* analysis which considers the distribution of all point-to point distances in a point pattern (Dixon, 2002), thus can be used to indicate the intensity, direction, and scale of pattern.

Ripley's *K* function is based on the number of points tallied within a given distances:

$$K(d) = \lambda^{-1}E(d)$$

where $E(d)$ is the expected number of points within a distance d from an arbitrary point and λ is estimated by point density.

$$\hat{K}(d) = \lambda^{-1}\sum_{i=1}^{n}\sum_{j=1}^{n}[\delta_{ij}(\|xi - xj\| \leq d)]/n$$

for $i \neq j$, where is an indicator function that takes on a value of 1 if the condition is true, else 0. The condition is that the Euclidian distance between points is less than or equal to d. That is, $\hat{K}(d)$ is a tally of the cumulative proportion of points comprised in each interval of d.

For completely random data, the expected value of $K(d) = \pi d^2$. $K(d)$ is transformed to

$$\hat{L}(d) = \sqrt{\frac{K(d)}{\pi}}$$

which produces a plot of $\hat{L}(d)$ against d where the expectation under randomness is a line with a slope of 1, or $L(d) = d$.

Ripley derived approximations of the test of significance depend on an assumption of normality. Because the actual distribution is unknown and assumption about normality could be under edge effect, in practice, Ripley's *K* function is generated from the test data. The data are then randomized to generate the test of significance as confidence limits. ESRI ArcGIS 10 was used to run Ripley's *K* analysis on birds observed within 37 100-m radius circles surrounding the bird census points. Spatial distribution pattern was determined by the visual interpretation of ESRI ArcGIS output plots. To assess temporal changes in bird distribution pattern, Ripley's *K* analysis was performed for each sampling season.

The spatial distribution pattern of local breeding birds was assessed at two spatial scales. First, bird distribution pattern was assessed over a range of distances within stand level using Ripley's *K* analysis for local breeding birds observed during three sampling seasons (early breeding season, migration, and peak breeding season). Stand size referring simply to 100 m radial circles surrounding the bird census point were used. Computation was carried out by the spatial statistics function of ESRI ArcGIS10. To minimize edge effects that can seriously affect distance-based statistics, ArcGIS10 simulates outer boundary values. This method creates points outside the study area boundary that mirror those points found inside the boundary in order to correct for underestimates near the edges. Points that are within a distance equal to the maximum distance band of an edge of the study area are mirrored. The mirrored points are used so that edge points will have more accurate neighbor estimates.

At landscape level, directional variogram analysis was applied to determine whether each bird species had any directional trend in their spatial distribution pattern. For this analysis, bird total abundance throughout the season for each species was used.

3.3.4.2 Semivariogram Analysis

Semivariance is a function that describes the correspondence between measurements taken on samples located some distance apart. Semivariance can be used in two applications: purely descriptive studies, in which the semivariogram (a plot of semivariance against distance class) is used to characterize the spatial structure of the data; and predictive applications in which the semivariogram is fitted to a theoretical model, parameterized, and used to predict the regionalized variable at points where no measurements are taken. Semivariograms summarize and provide useful visual representation of the spatial structure of the data, yet produce no statistical results to test hypothesis about the data.

A semivariogram measures the mean variability between two points as a function of their distance from each other. It is calculated by equation;

$$2\gamma(d) = \frac{1}{n_d}\sum_{i}\sum_{i} w_{ij}\,(y_i - y_j)^2$$

where j is a point at distance d from i, and n_d is the number of points at that distance and the weights w_{ij} are an indicator function set to 1 if the pair of points is within the distance class. The summation term is a distance measure as well as a typical variance term. Dividing this by 2, makes it semi (half) variance, γ, which is how it is expressed. For geostatistics, this equation is often denoted as

$$\gamma(d) = \frac{1}{2(n-d)}\sum_{i}^{n-d}(y_{i+d} - y_i)^2$$

where summation is over all (n-d) pairs of points in lag distance class d.

As the behavior of semivariograms may differ in relation to direction, semivariograms were calculated for a range of directions. Experimental semivariograms were computed for lag based on the bird sample structure in the 0°, 45°, 90°, and 135° directions with the lag distance of 500 m. If the variance characteristics of $\gamma(d)$ are independent of the spatial direction, then $\gamma(d)$ is called isotropic; if not, then $\gamma(d)$ is called anisotropic (Legendre & Fortin, 1989).

3.3.5 Environmental Variables Affecting Community Structure across Habitat Types

NMDS is informative to assess differences in bird assemblages across habitat types, but is in adequate to investigate relationship between bird species and habitat characteristics (environment). For the correlation between bird species composition and environmental variables, which are affecting variation in bird community assemblages across habitat types, canonical correspondence analysis (CCA) was used. CCA is a technique to include environmental variables as part of the mathematical computations. Previous studies have demonstrated that CCA, which performs a least-square regression of plot scores (species' weighted averages) as dependent variables on environmental variables as independent variables, outperforms other ordination techniques such as NMDS (Gotelli & Ellison, 2013; Barbour et al., 1999). CCA produces plot and species location in ordination space and generates environmental vectors originating from the center. Further, the length of these vectors represents the gradient length of an environmental variable. In other words, vector length is proportional to the importance of that gradient, leading to more important environmental variables represented by longer vectors. Since the first introduction by ter Braak (1986), CCA has been used in a wide range of ecological studies (Young & Peacock, 1992; Hoeinghaus, Winemiller, & Birdbaum, 2007).

Environmental variables used to measure habitat characteristics (Appendix F) were calculated based on forest patch metrics derived from LiDAR-hyperspectral combined data, thus high multicollinearity among variables were inevitable. For this analysis, only statistically significant and non-redundant variables were retained. Statistical significance was determined by running analysis of variance (ANOVA) on CCA model. Environmental variables were added manually using 1000 Monte Carlo simulation. Non-redundancy was examined by checking variance inflation factors (VIF). Statistical significance was determined by p-values smaller than 0.01 and all variables with VIF greater than 10 were considered redundant. Additionally, because

CCA is sensitive to rare species and my census results included some chance observation, total abundance of local breeding birds was used for this analysis.

3.3.6 Minimum Spanning-Trees

To investigate when secondary forests start to have a habitat function for forest interior birds, minimum spanning tree (MST) analysis was performed. MST is an edge-weighted graph where weights are associated with each edge. MST finds a subset of the edges that forms a tree that includes every vertex, where the total weight of all the edges in the tree is minimized (Oksanen, 2013). Unlike a dendrogram which shows the distance between clusters, MST displays where actual points are joined. MST is generated based on the distance between points is the shortest. To measure distance between points (sites), Bray-Curtis dissimilarity index was employed.

3.3.7 Spatial Effects

Spatial structure is a fundamental component of ecosystems. The observed distribution of organisms may be caused by spatial dependence in response variables, or by the dependence of response variables on explanatory variables which themselves may be spatially structured (Borcard, Legendre, & Drapeau, 1992). In other words, if environmental variables selected to explain variations in bird species composition across habitat types had any underlying spatial pattern or bird distribution itself has any spatial structuring due to biotic interactions, it could create an issue of double counting spatial effects. Since Borcard et al. (1992) proposed a method based on partial CCA, increasing attention has been paid to incorporate the spatial structure of the response variables and explanatory variables. Partial CCA factors out the spatial component of the ecological variation by the following four variations: non-spatial environmental variation, spatially structured environmental variation, spatial variation that is not share by environmental variables, and non-spatial variation (unexplained variation) (Borcard & Legendre, 1992). This variation partitioning method has been applied to vegetation distribution (Borcard & Legendre, 1992; Hobson, Kirk, & Smith, 2000), bird distribution derived from Atlas data (Titeux et al., 2004). This study attempted to partial CCA to remove spatial component of the variation by applying CCA to a set of selected environmental variables which included geographic coordinate information of each census point.

3.3.8 Analysis on Foraging Guilds

Ecological communities contain groups of species which exploit resources in competitive ways. For avian communities, primary factors consisting of structures involve resource (food and space) availability, both of which entail competition. Previous bird foraging studies have demonstrated that the foraging trait is substantially flexible. For example, even for the same species, preference for foliage type or height can vary with time of day, weather, season, presence of competitors, presence of predators, or nesting period (Gill, 2006: Newton, 1998). High variability in foraging trait makes a foraging guild classification somewhat ambiguous, yet most members of the same foraging guild have similar habitat preferences. Examining how species in the same foraging guilds are related to forest stand structure at various scales can increase our knowledge for effective conservation planning.

To investigate how birds in the foraging guilds are related to selected environmental variables, 65 local breeding bird species were classified into 16 groups, based on the species specific foraging traits such as habitat requirements (aquatic, arboreal, ground...) and basic behaviors (Table 4.14). Classification of birds into foraging guilds was carried out based on behavior and habitat descriptions obtained from the Cornell Lab of Ornithology website.

CHAPTER 4

RESULTS AND DISCUSSION

4.1 Forest Cover Change Detection

Changes in land cover around Denton County between 1984 and 2006 were detected from Landsat imagery (*Figure 4.1*). Surface areas of forests converted to other land types are illustrated in Figure 4.2. This region lost 32% of forest cover in two decades (630 km^2 in 1984 and 431 km^2 in 2006). Construction of Lake Ray Roberts caused raised water conservation pool level of Lake Lewisville. Urban developments were major contributors to the loss of forests, making up approximately 40% of all forest loss. It is assumed that forests lost to the impoundment of the two reservoirs consisted mainly of bottomland hardwood forests.

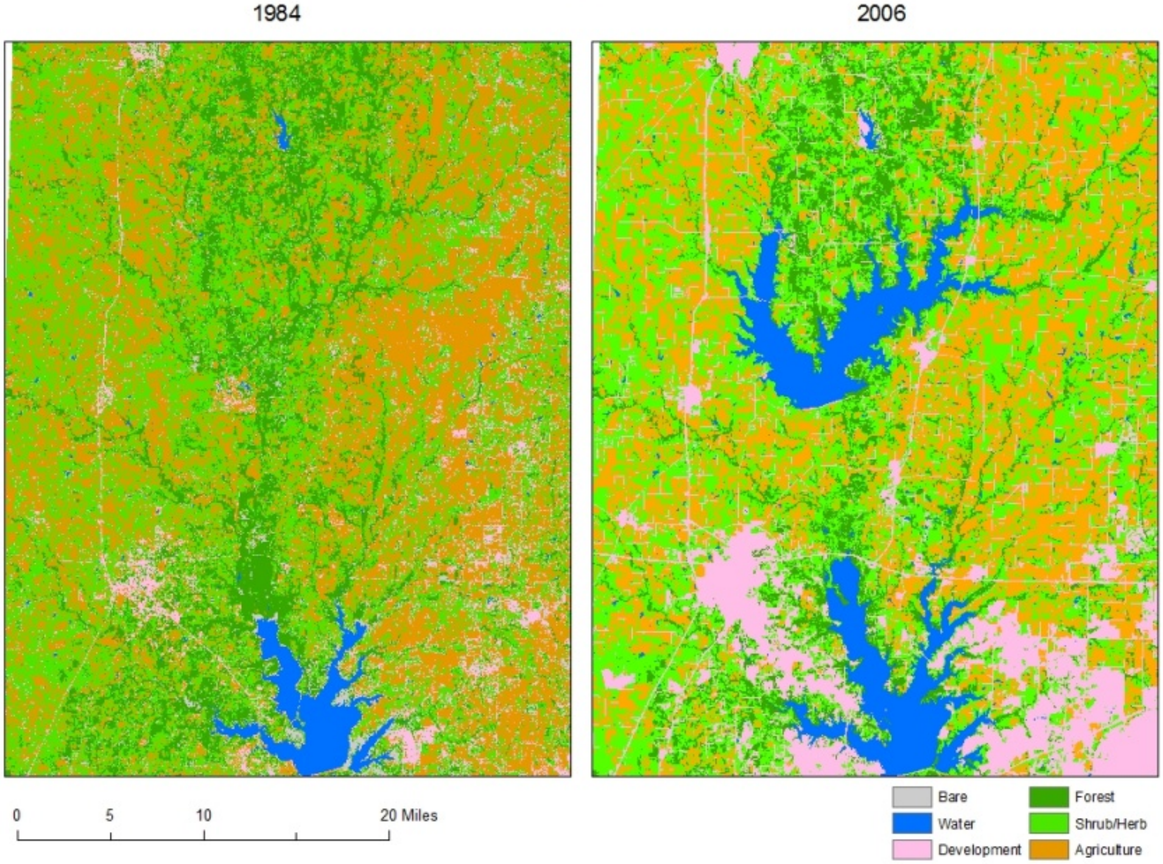


Figure 4.1. Forest changes around the study site between 1984 and 2006.

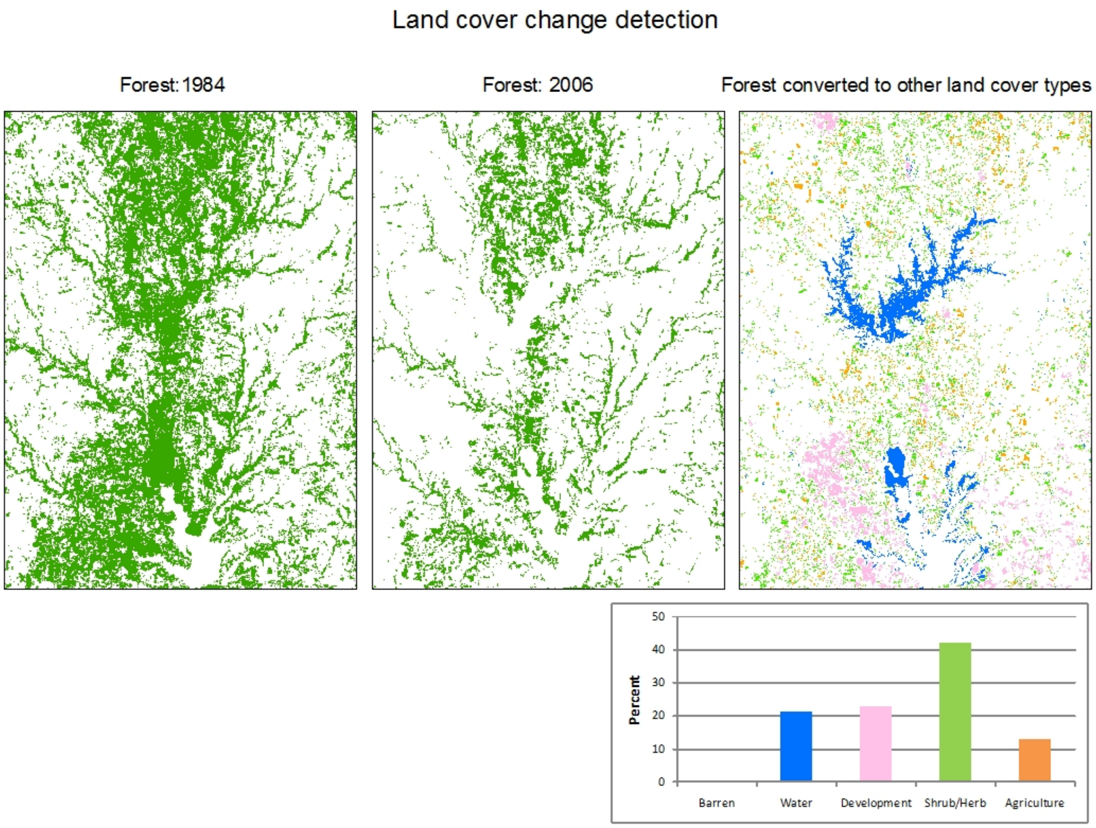


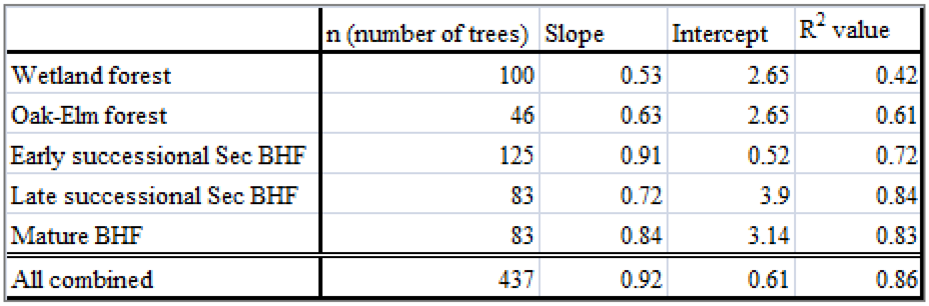
Figure 4.2. Relative percentage of forest area change around the study site between 1984 and 2006.

4.2 Accuracy Assessment of LiDAR Data

Accuracy assessment results on LiDAR-derived vegetation height are summarized in Table 4.1. Height was measured for a total of 437 trees that grow within 25×25 m quadrat plot at seven forested sites. There was one plot established in wetland forests, one site in oak-elm forests, one in secondary bottomland forests which is still in early successional stage, two secondary bottomland forest sites in late successional stage, and two in mature bottomland hardwood forests. A list of canopy tree and shrub species and associated properties are summarized in Table 4.2. Canopy species richness was the highest in mature bottomland hardwood forests and secondary forests in late successional stage and the lowest in wetland forest where 96% of trees were dominated by swamp privet (*Forestiera acuminata*) and green ash.

Table 4.1

Simple linear regression models examining the relationship between LiDAR vegetation height and the ground reference vegetation height. Data were collected from seven 25×25 m forested sites from five forest types: Wetland forest, Oak-Elm forest, Early successional secondary bottomland forest (Sec BHF), Late successional secondary bottomland forest, and Mature bottomland hardwood forest (Mature BHF).

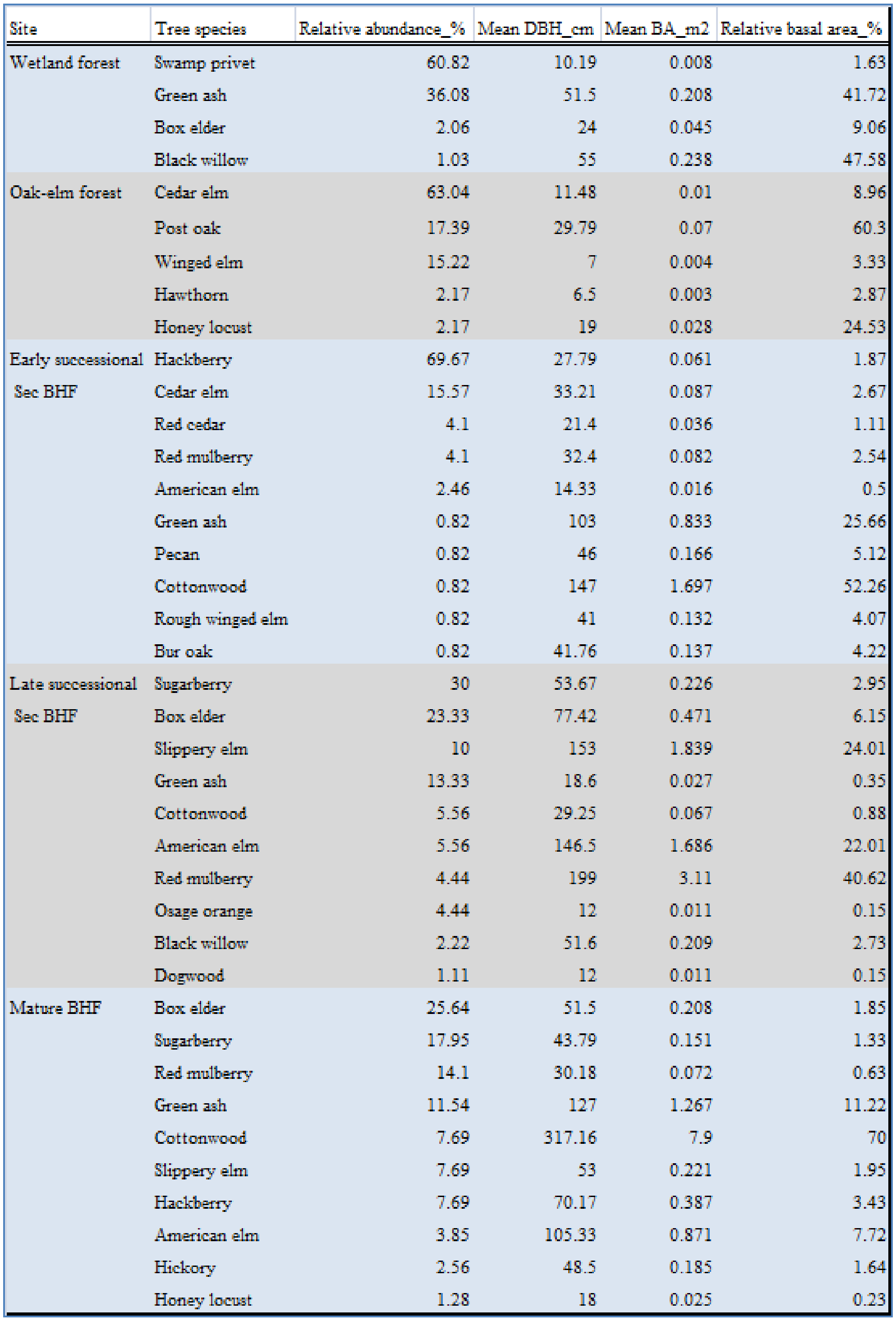


	n (number of trees)	Slope	Intercept	R^2 value
Wetland forest	100	0.53	2.65	0.42
Oak-Elm forest	46	0.63	2.65	0.61
Early successional Sec BHF	125	0.91	0.52	0.72
Late successional Sec BHF	83	0.72	3.9	0.84
Mature BHF	83	0.84	3.14	0.83
All combined	437	0.92	0.61	0.86

Higher species richness was observed in bottomland forests in different age cohorts, as compared to wetland forests and upland oak-elm forests. Although species richness was the same in mature bottomland hardwood forests and secondary forests in different successional stages, various relative abundance and relative basal area indicate differences among forests in different age cohorts. Early successional forest was dominated only by hackberry in terms of relative abundance, yet both late successional and mature bottomland hardwood forests were dominated by five species: Box elder, sugarberry (*Celtis laevigata*), red mulberry (*Morus rubra*), slippery elm (*Ulmus rubra*), and green ash. High similarities between early successional forest and mature bottomland hardwood forests in relative basal area suggest that there exist some uncut mature trees (such as cottonwood and green ash) in early successional stands, indicating difficulties in grouping stands in discrete age classes.

Table 4.2

Canopy tree species, relative abundance, mean DBH, and mean BA, and relative BA. Data were collected from seven 25×25 m forested site: wetland forest, oak-elm forest, early successional secondary bottomland forest (Sec BHF), late successional secondary bottomland forests, mature bottomland hardwood forest (Mature BHF).



Site	Tree species	Relative abundance_%	Mean DBH_cm	Mean BA_m2	Relative basal area_%
Wetland forest	Swamp privet	60.82	10.19	0.008	1.63
	Green ash	36.08	51.5	0.208	41.72
	Box elder	2.06	24	0.045	9.06
	Black willow	1.03	55	0.238	47.58
Oak-elm forest	Cedar elm	63.04	11.48	0.01	8.96
	Post oak	17.39	29.79	0.07	60.3
	Winged elm	15.22	7	0.004	3.33
	Hawthorn	2.17	6.5	0.003	2.87
	Honey locust	2.17	19	0.028	24.53
Early successional Sec BHF	Hackberry	69.67	27.79	0.061	1.87
	Cedar elm	15.57	33.21	0.087	2.67
	Red cedar	4.1	21.4	0.036	1.11
	Red mulberry	4.1	32.4	0.082	2.54
	American elm	2.46	14.33	0.016	0.5
	Green ash	0.82	103	0.833	25.66
	Pecan	0.82	46	0.166	5.12
	Cottonwood	0.82	147	1.697	52.26
	Rough winged elm	0.82	41	0.132	4.07
	Bur oak	0.82	41.76	0.137	4.22
Late successional Sec BHF	Sugarberry	30	53.67	0.226	2.95
	Box elder	23.33	77.42	0.471	6.15
	Slippery elm	10	153	1.839	24.01
	Green ash	13.33	18.6	0.027	0.35
	Cottonwood	5.56	29.25	0.067	0.88
	American elm	5.56	146.5	1.686	22.01
	Red mulberry	4.44	199	3.11	40.62
	Osage orange	4.44	12	0.011	0.15
	Black willow	2.22	51.6	0.209	2.73
	Dogwood	1.11	12	0.011	0.15
Mature BHF	Box elder	25.64	51.5	0.208	1.85
	Sugarberry	17.95	43.79	0.151	1.33
	Red mulberry	14.1	30.18	0.072	0.63
	Green ash	11.54	127	1.267	11.22
	Cottonwood	7.69	317.16	7.9	70
	Slippery elm	7.69	53	0.221	1.95
	Hackberry	7.69	70.17	0.387	3.43
	American elm	3.85	105.33	0.871	7.72
	Hickory	2.56	48.5	0.185	1.64
	Honey locust	1.28	18	0.025	0.23

The relationship between LiDAR-derived vegetation height and ground truth data for six forest types (wetland forest, oak-elm forest, early successional bottomland hardwood forest, late successional bottomland hardwood forest, and mature bottomland hardwood forest) was examined by performing simple linear regression analysis (

Figure 4.3).

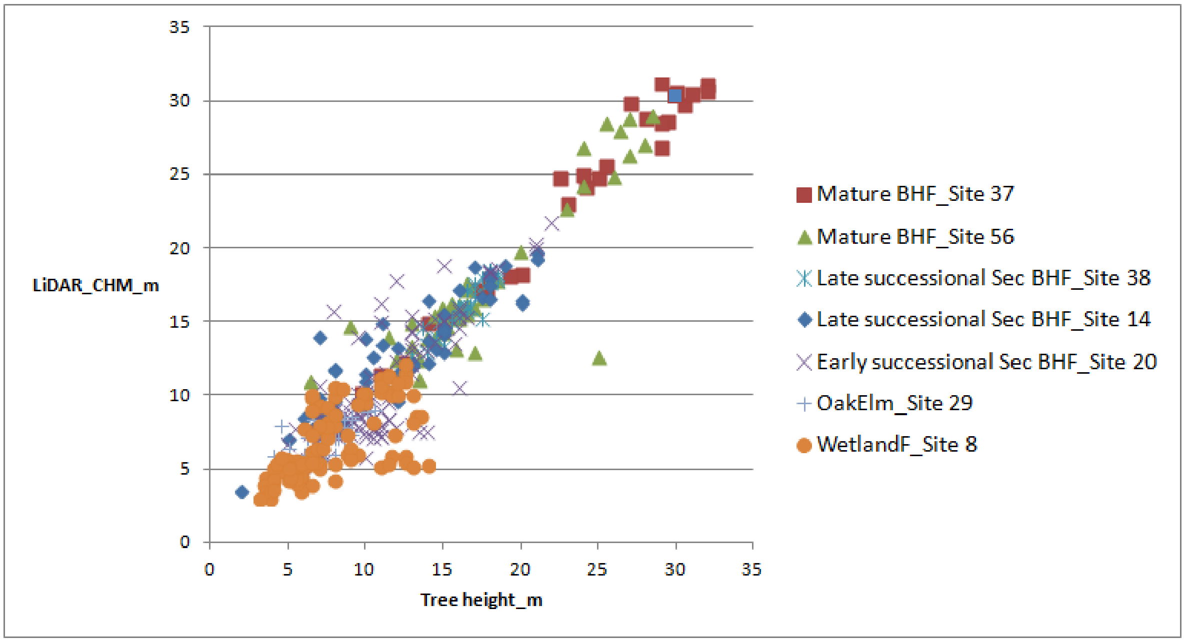


Figure 4.3. Linear regression showing the relationship between LiDAR derived vegetation height and ground reference height for seven forested sites from five forest types: Wetland forest, Oak-Elm forest, Early successional secondary bottomland forest (Sec BHF), Late successional secondary bottomland forest, and Mature bottomland hardwood forest (Mature BHF).

There were positive correlations between the LiDAR height and the ground truth data in all forest types. The largest slope (0.91) and the smallest intercept (0.52) of early successional forest indicate its homogeneous stand vertical structure where most trees in similar height create high density stands. Mature bottomland hardwood forests and late successional secondary forest had similar vertical heterogeneity, described by slope values of 0.84 and 0.72, intercept of 3.14 and 3.9, R^2 values of 0.83 and 0.84, respectively. Oak-elm forest where no known agriculture occurred showed similar stand structure with mature bottomland forests and late successional secondary forests, although the R^2 value (0.61) was quite low. Wetland forests which consists mainly of swamp privet and few tall trees such as green ash, box elder, and black willow had relatively low fit (slope = 0.53, $R^2 = 0.42$). It appears that both stand density and vegetation surface structure affected the accuracy in LiDAR height measurements. Relatively dense and structurally homogeneous stands had a higher possibility to be accurately measured by LiDAR. In dense stands, if the LiDAR laser missed the canopy top, it had a higher probability to hit adjacent trees, resulting in relatively high accuracy. On the other hand, vegetation height derived from LiDAR in sites sparsely covered by tall trees or largely heterogeneous stands tended to be underestimated.

4.3 Hyperspectral Data Analysis

Hyperspectral reflectance data were examined for several sites across the study site. As shown in Figure 4.4, spectral reflectance signatures differentiated some characteristic vegetation communities: grass-herb communities with the highest reflectance and oak-elm forests with the lowest overall. To further delineate vegetation communities across the study site, PCA was performed on hyperspectral reflectance data (*Figure 4.4*).

PCA resulted in five components which contained 99.5 % of variability of the data. The first and the second PC explained 92% and 98.4% of variability. Resultant imagery (*Figure 4.4*) shows that although PCA could partially differentiate some distinctive features such as water, wetlands, grass-herb, and some areas of mature riparian forests sufficiently, the ability to distinguish deciduous forests by dominant species or by various successional stages was limited. There were two major reasons for this limited detestability. First, the spatial resolution of this imagery was 30×30 m wherein many different plants coexist.

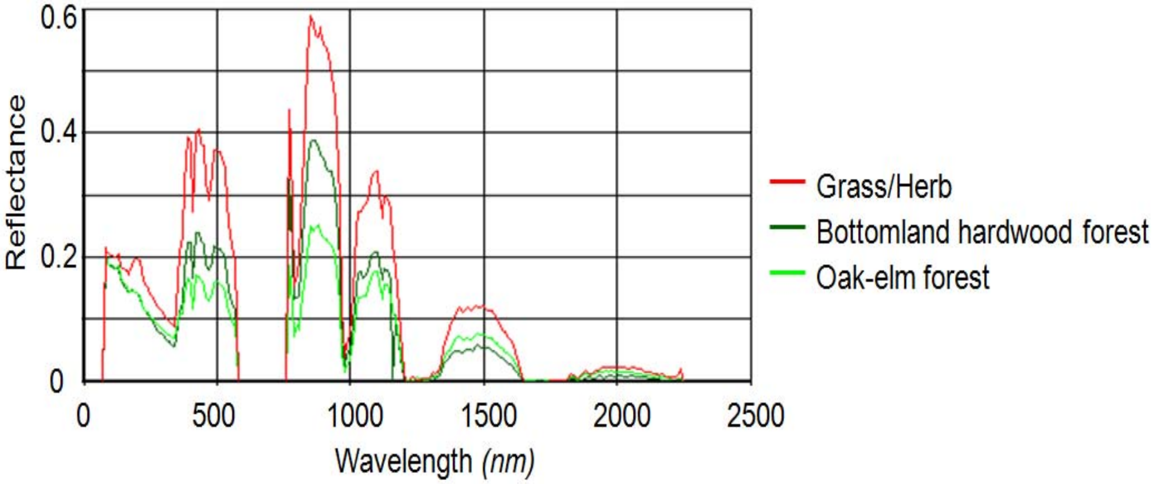


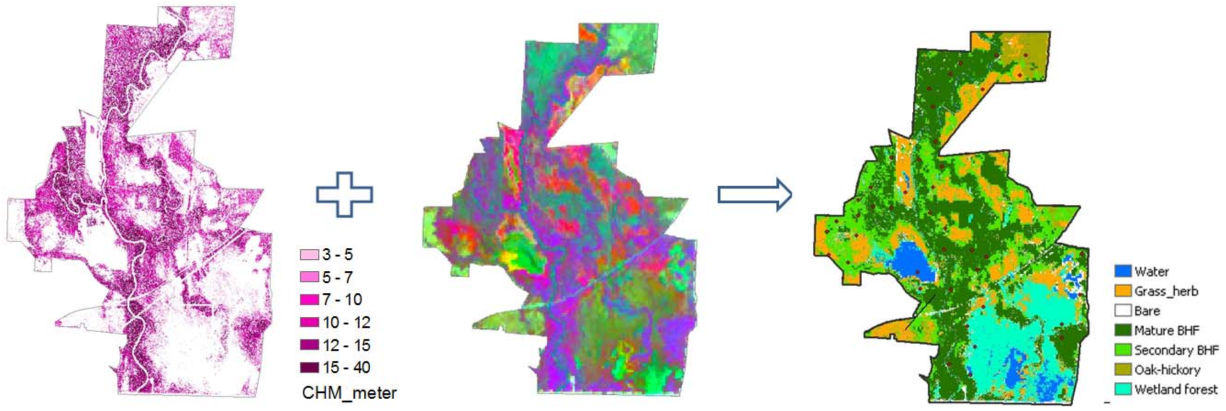
Figure 4.4. Spectral reflectance curves showing different plant communities.

Previous studies have demonstrated that a spectral unmixing technique can effectively delineate multiple objects occupying a pixel based on spectral characteristics of the target objects. For example, a single pixel that is composed of 50% water, 25% bare soil, and 25% vegetation can be classified to each class separately based on the relative abundance instead of classified to one mixed pixel. Then a linear spectral unmixing technique can be applied to match each vegetating type with the specific reflectance. Linear spectral unmixing is a sub pixel classification method for remote sensing used to determine the relative proportions of materials that are depicted in hyperspectral imagery based on the materials' spectral characteristics (Jensen, 2005). Spectral reflectance of a variety of materials including vegetation is available at the USGS digital spectral library provides.

Despite the availability of spectral reflectance information for various materials and plant species, the identification of unique plant species is difficult. Hyperspectral reflectance signatures are proven to be useful to identify broad categories of forest cover such as conifer vs. deciduous forests (Martin et al., 1998). Hundreds of continuous narrow bands of hyperspectral data are capable of detecting subtle differences. However, plant species identification is limited by numerous factors such as the season of data collection, atmospheric properties, plant community composition, or other site specific conditions which can affect plant reflectance, absorption, and transmittance. Difficulties in separating spectral reflection of vegetation in bottomland hardwood forests multiply when most canopy species are deciduous and have similar spectral reflectance signatures. The objective of incorporating hyperspectral data for bird habitat assessment, therefore, is to differentiate broadly defined plant communities. Regardless of relatively coarse spatial resolution of the available hyperspectral data, spectral reflectance signatures derived from hyperspectral imagery could still be an informative descriptor to delineate different vegetation cover based on dominant plant species occupying the pixel.

4.4 Accuracy Assessment of the Habitat Types Derived from LiDAR-Hyperspectral Integrated Data

Hyperspectral-derived five PCs and LiDAR derived vegetation height were integrated and classified to seven land cover types using a supervised classification method and maximum likelihood algorithm (*Figure 4.5*). No classification map is free of errors, thus classification accuracy was assessed by verifying the habitat types of 200 randomly generated points based on the field knowledge. The relationship between LiDAR-hyperspectral integrated imagery which was classified to six habitat classes and the ground reference information was summarized in an error matrix. Overall classification accuracy is derived from confusion matrices by first counting how many pixels were classified the same on the ground and the map, and then dividing this by the total pixel numbers. Overall classification accuracy of five PCs derived from hyperspectral imagery only was 72.5% (Table 4.3) and LiDAR-hyperspectral integrated imagery was 82.5% (Table 4.4), leading to an increase in classification accuracy by 10 %.



Canopy height derived from LiDAR

Plant communities derived from Hyperspectral data

LiDAR and Hyperspectral data combined

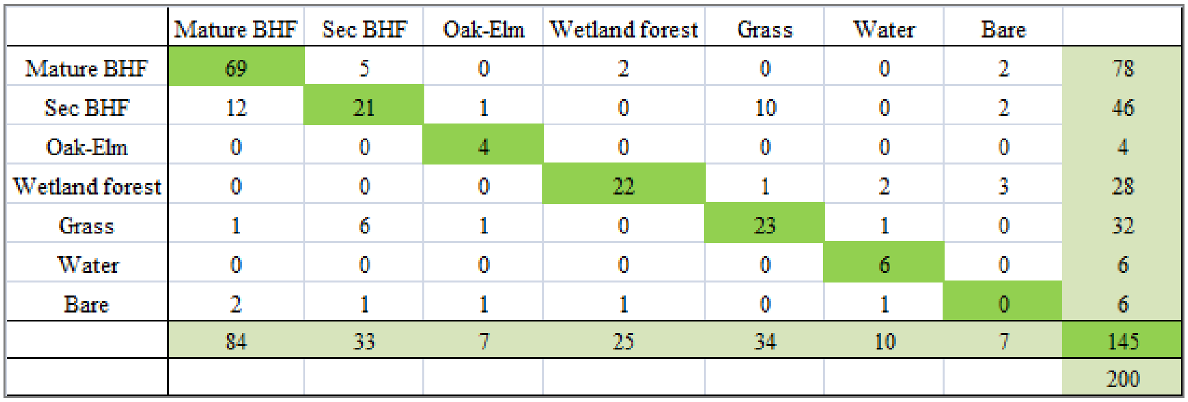
Figure 4.5. Canopy height derived from LiDAR, plant communities derived from hyperspectral data, and habitat types classified from LiDAR-hyperspectral integrated data.

A drawback of the overall classification accuracy is that this measure doesn't provide any information about how well individual classes were classified. To examine the classification performance on each class, confusion matrices were produced for hyperspectral only and LiDAR-hyperspectral map. The producer's accuracy and the user's accuracy are widely used measures of class accuracy. The user's accuracy (an error of commission) refers to the percentage of extra pixels in a class, whereas the producer’s accuracy (an error of omission) refers to the probability that a certain land type is classified as such (Jensen, 2005).

The producer's and the user's accuracy corresponding to each land cover type derived from five PCs of hyperspectral data is summarized in Table 4.3 and LiDAR-hyperspectral integrated data is summarized in Table 4.4. An estimate of the producer's accuracy of mature bottomland hardwood forests derived from hyperspectral image only was 82.14%, while the user's accuracy was 88.46%. These high values indicate that both classification accuracies were quite high for mature bottomland forests. Not all classes generated similarly high accuracies, however. An extreme case was found in bare fields, with both classification accuracies of 0.

Table 4.3

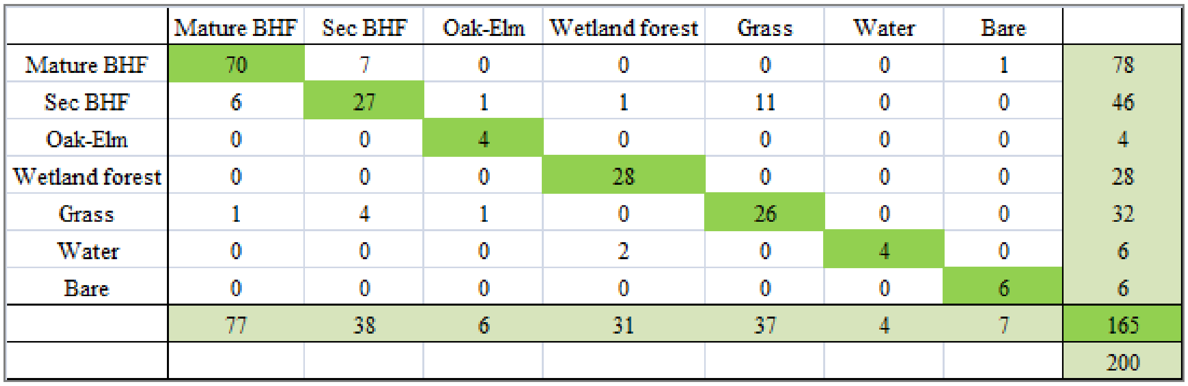
Error matrix for five PCs of hyperspectral data classified to six land cover types: mature bottomland hardwood forest (Mature BHF), secondary bottomland hardwood forest (Sec BHF), oak-elm forest (Oak-Elm), wetland forest, grass-herb (Grass), water, and bare field (bare).



	Mature BHF	Sec BHF	Oak-Elm	Wetland forest	Grass	Water	Bare	
Mature BHF	69	5	0	2	0	0	2	78
Sec BHF	12	21	1	0	10	0	2	46
Oak-Elm	0	0	4	0	0	0	0	4
Wetland forest	0	0	0	22	1	2	3	28
Grass	1	6	1	0	23	1	0	32
Water	0	0	0	0	0	6	0	6
Bare	2	1	1	1	0	1	0	6
	84	33	7	25	34	10	7	145
								200

Table 4.4

Error matrix for LiDAR-hyperspectral integrated data classified to six land cover types: mature bottomland hardwood forest (Mature BHF), secondary bottomland hardwood forest (Sec BHF), oak-elm forest (Oak-Elm), wetland forest, grass-herb (Grass), water, and bare field (bare).

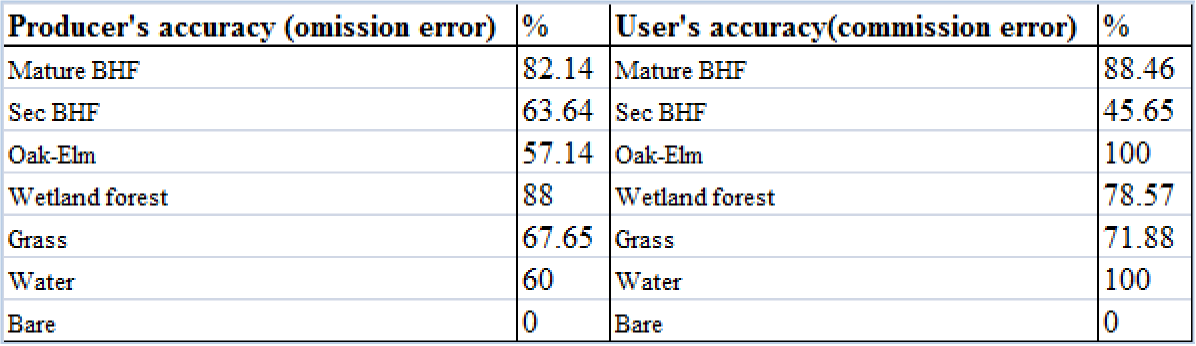


	Mature BHF	Sec BHF	Oak-Elm	Wetland forest	Grass	Water	Bare	
Mature BHF	70	7	0	0	0	0	1	78
Sec BHF	6	27	1	1	11	0	0	46
Oak-Elm	0	0	4	0	0	0	0	4
Wetland forest	0	0	0	28	0	0	0	28
Grass	1	4	1	0	26	0	0	32
Water	0	0	0	2	0	4	0	6
Bare	0	0	0	0	0	0	6	6
	77	38	6	31	37	4	7	165
								200

Another example was found in water sites, with the producer's accuracy of 60% and the user's accuracy of 100%. This indicates that for a user, 100% of all pixels classified as water were indeed water in the field. As producer, however, only 60% of water was correctly classified. Disparities between the producer's accuracy and the user's accuracy were most prominent in secondary forests, oak-elm forests, and bare fields (Table 4.5). When height information derived from LiDAR was integrated, classification accuracies increased for most land cover types. Particularly, height information was useful to delineate vegetated vs. non vegetated pixels, i.e., water and bare fields (Table 4.6). Inclusion of height also contributed a better classification for forested sites occupied by plants with specific height characteristics such as mature bottomland hardwood forests, wetland forests, and grass-herbaceous sites.

Table 4.5

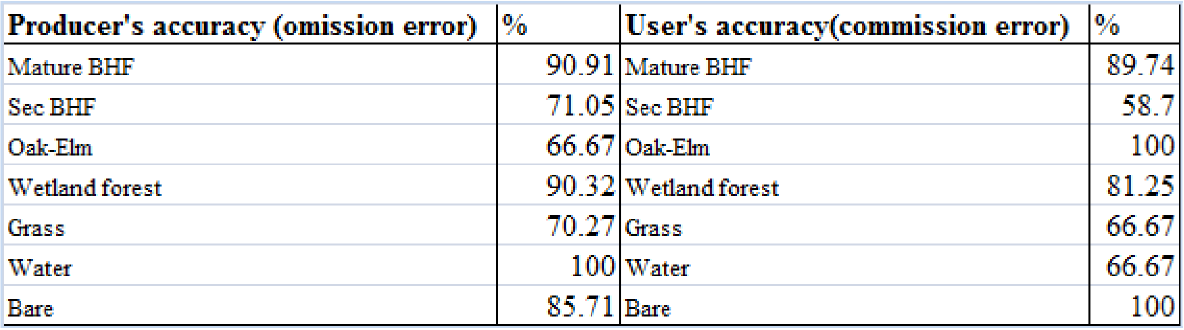
Producer's and user's accuracy of supervised classification using five PCs of hyperspectral data classified to six land cover types: mature bottomland hardwood forest (Mature BHF), secondary bottomland hardwood forest (Sec BHF), oak-elm forest (Oak-Elm), wetland forest, grass-herb(Grass), water, and bare ground (bare).



Producer's accuracy (omission error)	**%**	**User's accuracy(commission error)**	**%**
Mature BHF	82.14	Mature BHF	88.46
Sec BHF	63.64	Sec BHF	45.65
Oak-Elm	57.14	Oak-Elm	100
Wetland forest	88	Wetland forest	78.57
Grass	67.65	Grass	71.88
Water	60	Water	100
Bare	0	Bare	0

Table 4.6

Producer's and user's accuracy of supervised classification using LiDAR-hyperspectral integrated data classified to six land cover types: mature bottomland hardwood forest (Mature BHF), secondary bottomland hardwood forest (Sec BHF), oak-elm forest (Oak-Elm), wetland forest, grass-herb(Grass), water, and bare ground (bare).



Producer's accuracy (omission error)	**%**	**User's accuracy(commission error)**	**%**
Mature BHF	90.91	Mature BHF	89.74
Sec BHF	71.05	Sec BHF	58.7
Oak-Elm	66.67	Oak-Elm	100
Wetland forest	90.32	Wetland forest	81.25
Grass	70.27	Grass	66.67
Water	100	Water	66.67
Bare	85.71	Bare	100

Although the land cover type specific accuracy increased with an inclusion of LiDAR-derived vegetation height, there exists a possibility that the agreement between the classified map and the ground reference data were obtained by chance. Kappa statistics ($\hat{\kappa}$) were calculated for classification maps derived both from hyperspectral only and LiDAR-hyperspectral infused data.

Kappa statistics ($\hat{\kappa}$) or the classification map derived from five PCs of hyperspectral imagery was 62.8%. When vegetation height information derived from LiDAR was added, the kappa coefficient value increased to 76.7 %. Although both kappa statistics values are categorized to moderate accuracy range, approximately 14% increase was achieved by LiDAR-hyperspectral integrated data.

In summary, integrating LiDAR-derived vegetation height increased the overall classification accuracy (calculated from an error matrix) by 10% and kappa statistics value by 14%. There are many factors that could have affected this accuracy assessment results. First of all, spectral reflectance derived from hyperspectral imagery was obtained in summer, while LiDAR-derived vegetation height was acquired in spring when some trees have not fully developed their leaves. Although the accuracy assessment on LiDAR height generated a relatively high accuracy, the effect of the temporal mismatch is inevitable. Moreover, the spatial and temporal scales of two sources of imagery need to be considered. The spatial resolution of hyperspectral imagery (30×30 m) was much coarser than the LiDAR CHM (1×1 m), leading to a high degree of edge miss classification issues. Edge misclassification effect could be multiplied by potential noises associated with spectral reflectance of hyperspectral imagery, which was used to delineate different vegetation communities.

Noise reduction from hyperspectral imagery is another key to consider. For this study, the dimensionality of hyperspectral data was reduced by employing PCA. Although PCA is an effective tool for data compression, it is not free of noise issues, therefore, not always the most appropriate method for vegetation classification. Jensen (2005) discusses the power of minimized noise fraction (MNF) transformation, which segregates noise in the data. Once segregated, noise has unit variance and no band-to band correlation, thus can theoretically performs better to reduce dimensionality of hyperspectral data. MNF performs a sequence of two orthogonal transformations, first for whitening the noise and then apply a standard PCA. Theoretically, for low noise data, results derived from PCA and MNF would be similar. For data with high noise MNF yields better classification results. Another noise reduction method is segmented PCA, which groups highly correlated bands. Each group is then compressed individually by PCA, thus all correlated bands are de-correlated (Almeida, 2004; Jensen, 2005). Bhattacharjee (2006), compared these three classification methods to classify land use for the area including this study site, and obtained the best classification accuracy from segmented PCA. No difference was observed in classification accuracy from PCA and MNF. For this study, only PCA was performed due to the limited software availability. It is assumed that vegetation classification obtained from MNF or segmented PCA, when combined with vegetation height information derived from LiDAR, could significantly increase the classification accuracy.

4.5 Bird Census Results

4.5.1 Bird Abundance

4.5.1.1 All Birds Combined

A total of 5559 birds (2554 in 2009 and 3005 in 2010) comprised of 91 species were observed from two years census (Appendix C). Figure 4.6 shows average number of birds detected in 2009 by habitat type and by season, while Figure 4.7 shows the average number detected in 2010.

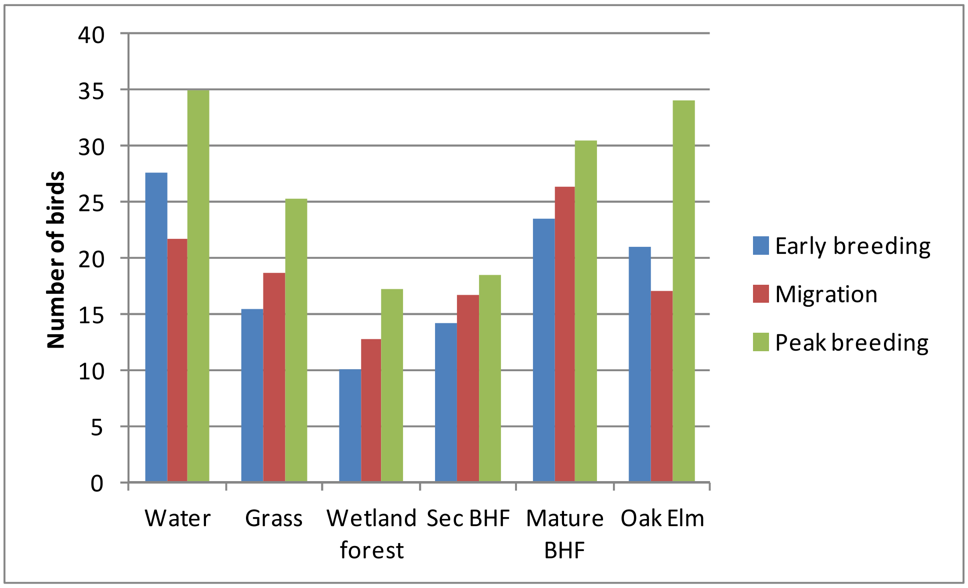


Figure 4.6. Average number of birds detected in 2009 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

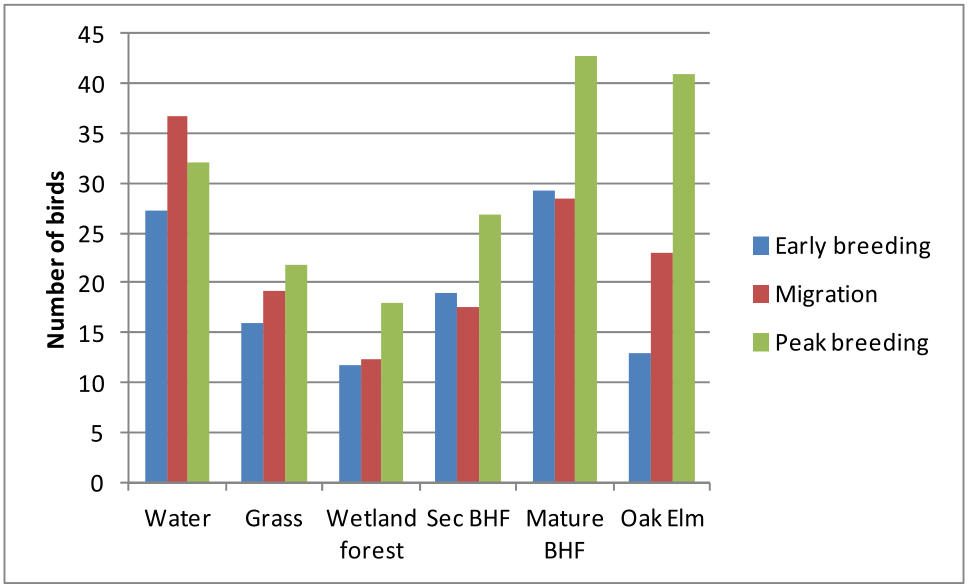


Figure 4.7. Average number of birds detected in 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

Because some of the census sites had issues ranging from GPS location errors to the difficulty to access to the point due to flooding conditions, only data collected at 37 out of 42 census points were used for this analysis. These sites were located as follows: three in water, four in grassland, four in wetland forest, nine in secondary forest, 16 in mature bottomland hardwood forest, and one in oak-elm forest. Because of unequal numbers of sites across habitat types as classified from the LiDAR-hyperspectral integrated map, bird abundance was averaged for each habitat type except for oak-elm forest which had only one site. Although the use of average may be inappropriate for this analysis, it still allows comparison of birds abundance detected across habitat types.

When aggregating all birds regardless of season, mature bottomland hardwood forest and water sites had the highest abundance in both census years. In 2009, the highest average number of birds was observed in water sites followed by mature bottomland hardwood forest. In 2010, the highest average number of birds was observed in mature bottomland hardwood forests followed by those observed in water. The lowest abundance was recorded in wetland forest. In 2009, abundance was the highest during the peak breeding season across all habitat types. This pattern was consistent in 2010 except for water site.

4.5.1.2 Local Breeding Birds vs. Others

To examine whether birds' breeding and migratory range affect the birds' abundance across habitat types, all birds were classified into two groups: local breeding birds and others. Local breeding birds include year-round residents and neo-tropical songbirds whose breeding range overlap with the study site. Others include groups of wintering birds and migratory flyovers who utilize the GBC site as wintering ground or resting sites during migration. Species classification was based on the range map created by Nature Serve and provided by the Cornell Lab of Ornithology (Cornell Lab of Ornithology, 2013). Local breeding birds occupied 94.1% and 86.8% of total birds observed in 2009 and 2010, respectively. Abundance of local breeding birds across habitat types had similar trend with total observed birds. In both census years, when all birds are combined, mature bottomland hardwood forests had the highest average numbers in 2009 (derived from *Figure 4.8*) and in 2010 (derived from *Figure 4.9*). Wetland forests had consistently low bird abundance, with respect to all birds observed birds in 2009 and 2010. The number of local breeding birds gradually increased as the season progressed across all habitat types.

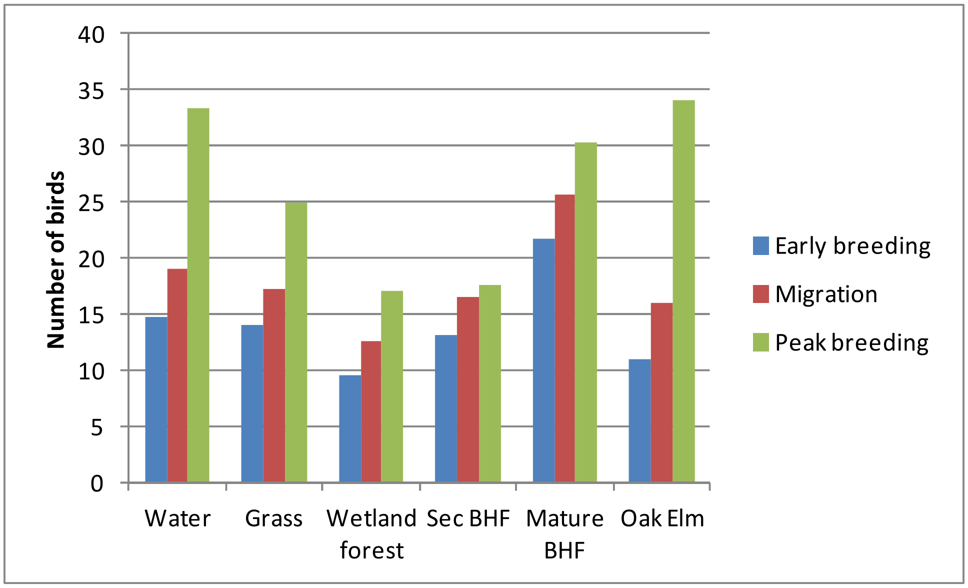


Figure 4.8. Average number of local breeding birds observed in 2009 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

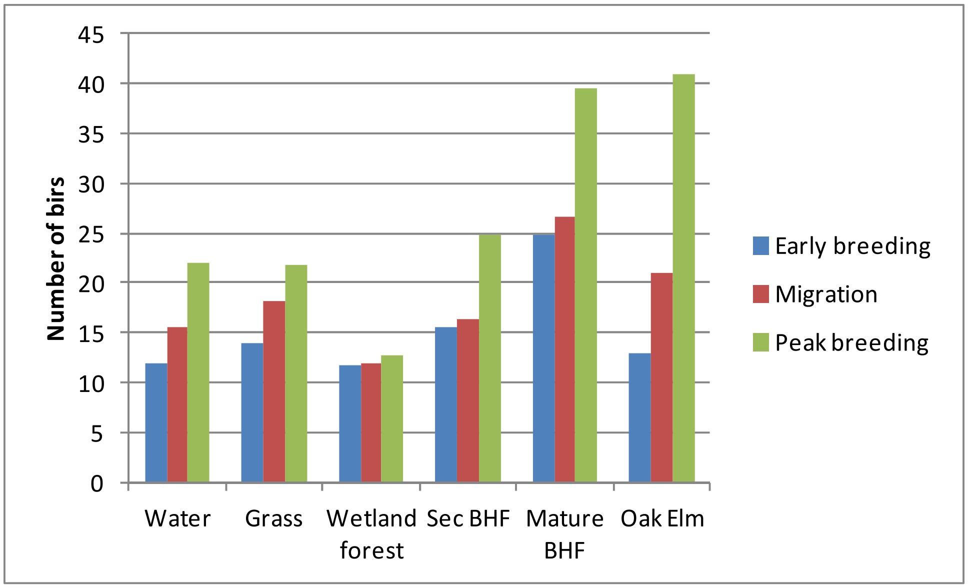


Figure 4.9. Average number of local breeding birds observed in 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

Numbers of the other type of bird group which included both wintering birds and migratory flyovers showed a slightly different pattern from local breeding birds. In 2009, 144 wintering-migratory flyovers were recorded. Of this, 46.2% were observed in water sites followed by 30.6% in mature bottomland hardwood forests and 29.4% in oak-elm forest. The lowest abundance was observed in wetland forest (2.7%). Water site attracted the highest average number of birds across all habitat types in 2010 as well. Of 389 wintering-migratory flyovers detected in 2010, 51.5% was observed in water site. In 2010, oak-elm forest (2.2%) and grassland (3.3%) had lower abundance in addition to wetland forest (6.1%), which recorded the lowest abundance in 2009. In 2009, the highest abundance was detected during early breeding season across all habitat types (*Figure 4.10*), whilst in 2010 this trend held true only for grassland and mature bottomland hardwood forest (*Figure 4.11*). Inconsistent temporal changes in bird abundance of this group suggest that the observation of migratory birds tends to be a chance event. Differences in non-local breeding birds detected in 2009 and 2010 could be attributed to weather conditions that influenced migration timing of the year.

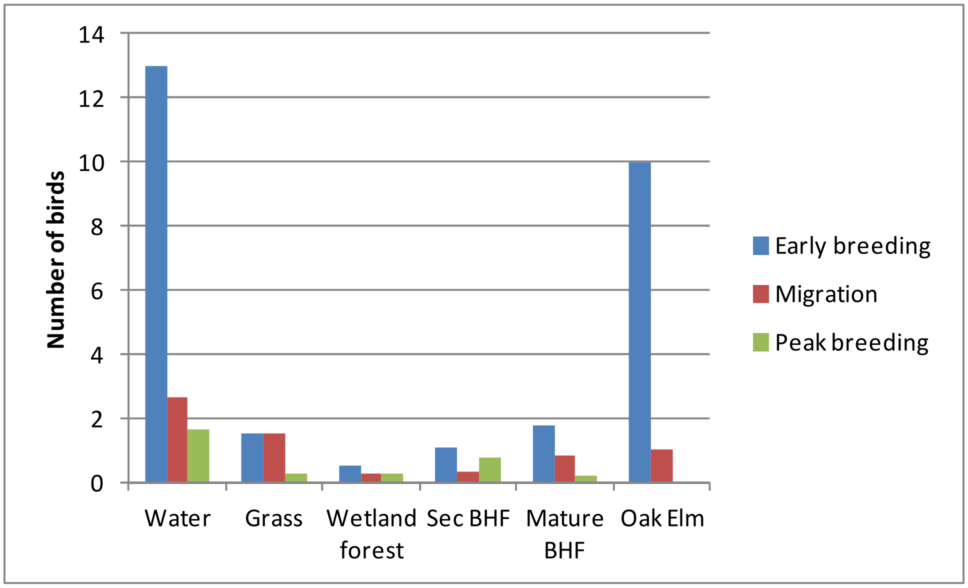


Figure 4.10. Average number of non-local breeding birds observed in 2009 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

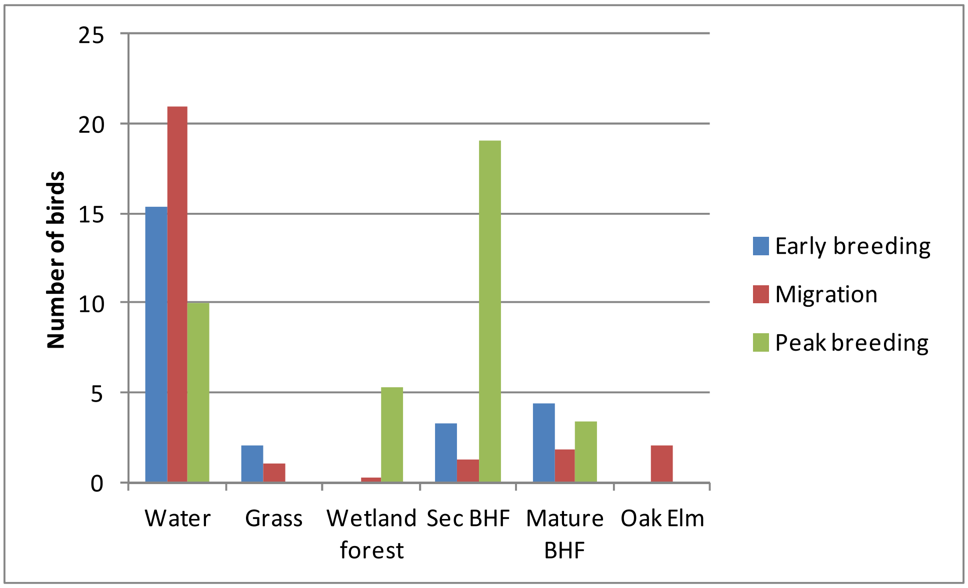


Figure 4.11 Average number of non-local breeding birds observed in 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

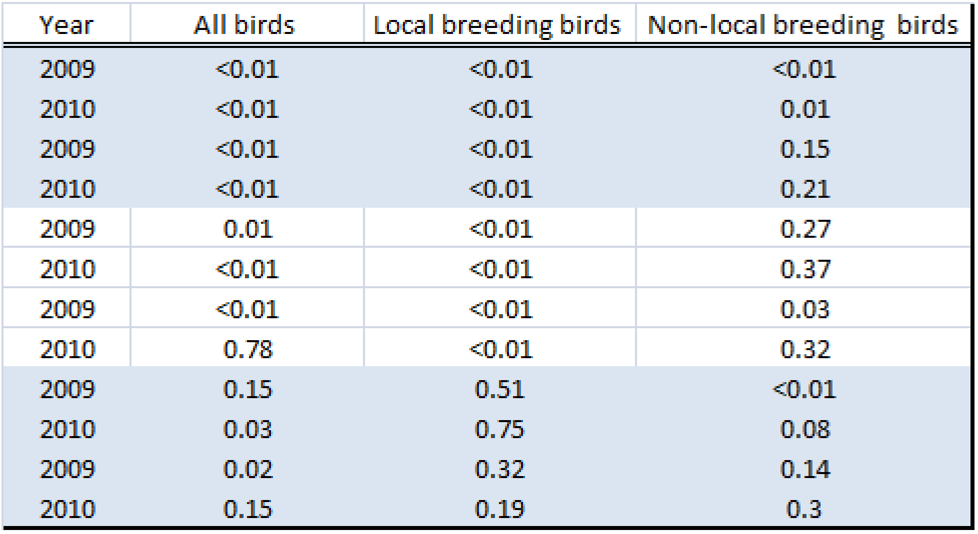
4.5.2 Community Indices

4.5.2.1 All Birds Combined

Species diversity was measured by Shannon-Wiener index (*H*) and Simpson's index (*D*). Species evenness was calculated from Pielou's evenness (*J*). Statistical significance of the effect of habitat types and sampling season on species diversity, richness, and evenness of local breeding birds and other communities, was tested by one-way analysis of variance (ANOVA) (Table 4.7).

Table 4.7

Summary of p-values of one-way ANOVA performed to analyze the effective of either habitat or habitat on Shannon-Wiener index (H), Simpson's index (D), and Pielou's evenness index (J) for 2009 and 2010 for all birds, local breeding birds, and non-local breeding birds.



Year	All birds	Local breeding birds	Non-local breeding birds
2009	<0.01	<0.01	<0.01
2010	<0.01	<0.01	0.01
2009	<0.01	<0.01	0.15
2010	<0.01	<0.01	0.21
2009	0.01	<0.01	0.27
2010	<0.01	<0.01	0.37
2009	<0.01	<0.01	0.03
2010	0.78	<0.01	0.32
2009	0.15	0.51	<0.01
2010	0.03	0.75	0.08
2009	0.02	0.32	0.14
2010	0.15	0.19	0.3

When all birds were combined, mature bottomland hardwood forests had the highest species diversity (*H*) across all habitat types in 2009 ($F_{5,105}$= 5.72, p-value < 0.01) and 2010 ($F_{5,105}$=9.75, p-value < 0.01). Tukey's multiple comparison's procedure (α=0.05) was then performed to identify which habitat type had a significantly higher diversity. Results suggested that species diversity (*H*) in mature bottomland was significantly higher than wetland forests and secondary forests in 2009 (*Figure 4.12*). In 2010, mature bottomland forests along with water and secondary forests had significantly higher diversity than wetland forest (*Figure 4.12*). Sampling season also affected species diversity across habitat types (*Figure 4.13*; $F_{2,\,108}$=6.39, p-value < 0.01). In addition, mature bottomland hardwood forests had higher diversity (*H*) than secondary forests. To identify which sampling season had a significant effect on species diversity, Tukey's multiple comparison procedure was performed. Although the increase in species diversity (*H*) from migration to peak breeding was minor, there was a steady increase in species diversity (*H*) throughout the sampling season, with the highest diversity during peak breeding season during both census years (*Figure 4.13*).

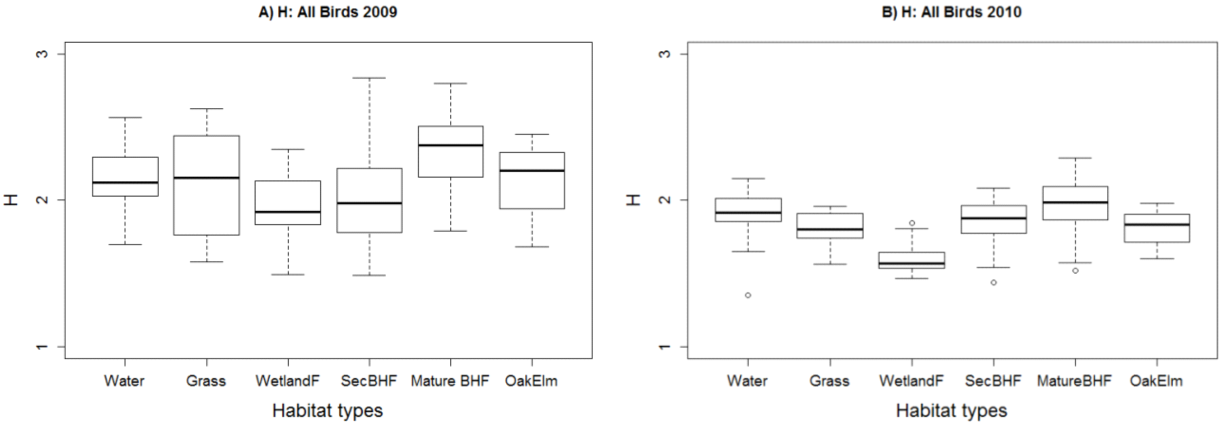


Figure 4.12. Shannon- Wiener diversity index (H) calculated for all birds across habitat types for A) 2009 and B) 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

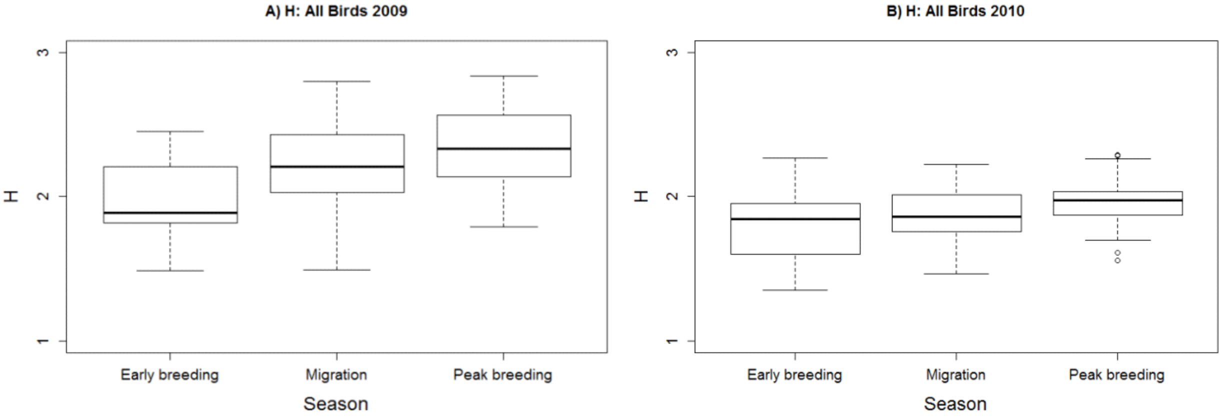


Figure 4.13. Temporal differences of Shannon-Wiener diversity index (H) calculated for all birds observed in A) 2009 and B) 2010.

Species diversity measured by Simpson's index (*D*) across habitat types in 2009 showed similar results with Shannon-Wiener index (*H*), with the highest diversity observed in mature bottomland hardwood forests, and the lowest diversity in wetland forests (*Figure 4.14*; $F_{5,105} = 3.05$, p-value < 0.01). Tukey's multiple comparison procedure ($\alpha=0.05$) detected a statistical significance between mature bottomland hardwood forests and wetland forests. In 2009, species diversity (*D*) increased throughout the season, with significant increase from early breeding to migration (p-value <0.01) and minor increase from migration to peak breeding season (p-value < 0.1). In 2010, however, Simpson's diversity (*D*) showed a slightly different pattern from the other species diversity index (*H*). Although the lowest diversity in wetland forest was similar to species diversity (*H*), the highest diversity occurred in water sites, followed by mature bottomland hardwood forests (*Figure 4.14*; $F_{5,105} = 3.84$, p-value < 0.01). In 2010, species diversity (*D*) did not significantly differ throughout the sampling season (*Figure 4.15*; $F_{2,108} = 0.24$, p-value = 0.78).

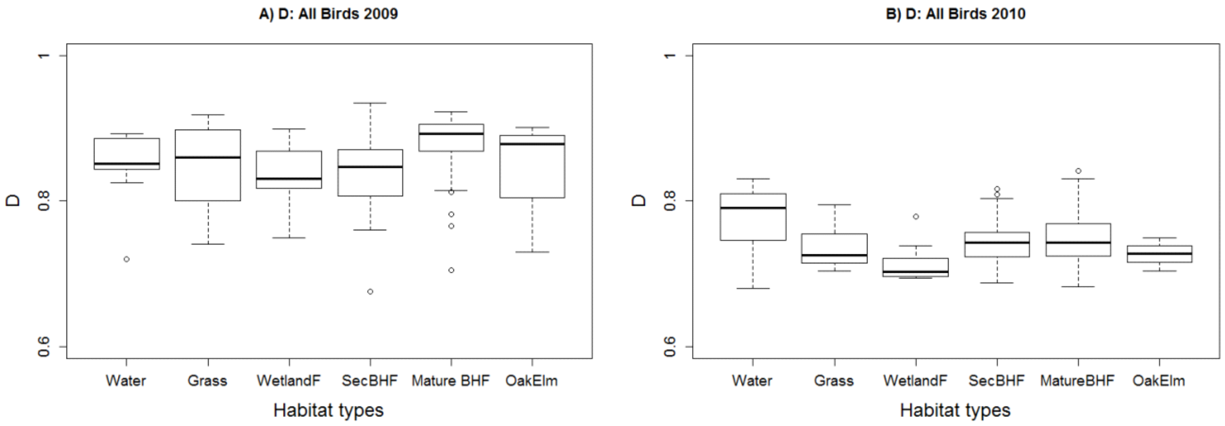


Figure 4.14. Simpson’s diversity index (D) calculated for all birds across habitat types for A) 2009 and B) 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

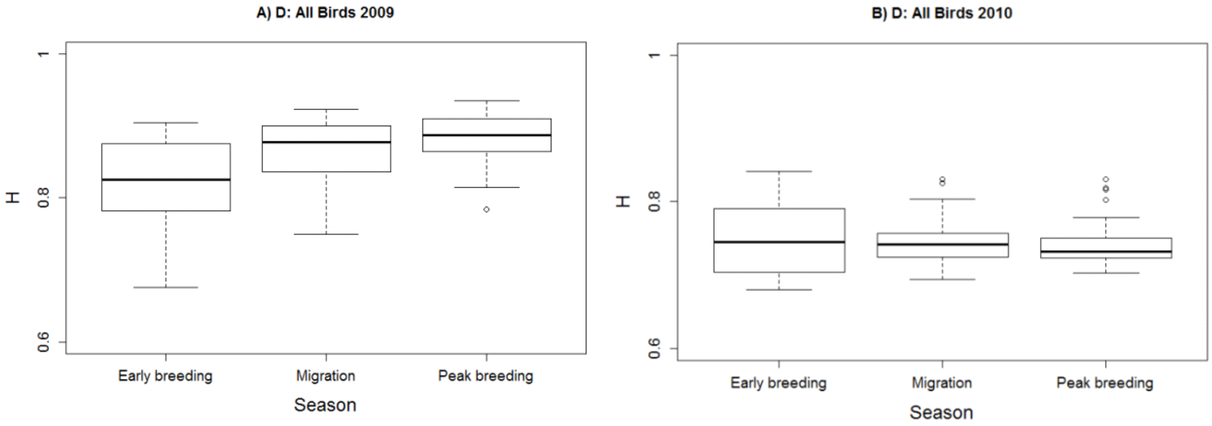


Figure 4.15. Temporal differences in Simpson's diversity index (D) for all birds observed in A) 2009 and B) 2010.

Evenness measured by Pielou's index (*J*) was unaffected neither by habitat types in 2009 (*Figure 4.16*; $F_{5,105} = 1.63$, p-value = 0.15). In contrast, wetland forests had the highest evenness across all habitat types in 2010 ($F_{5,105} = 3.04$, p-value = 0.01). Tukey's multiple comparison test ($\alpha = 0.05$) detected a statistical significance between wetland forests and mature bottomland forests in 2010 (p-value = 0.02). The sampling season had a minor but significant increasing trend in 2009 ($F_{2,108} = 3.75$, p-value = 0.02), whereas in 2010 a statistically significant declining trend was observed throughout the season and the results were statistically significant (*Figure 4.17*; $F_{2,108} = 25.74$, p-value < 0.01).

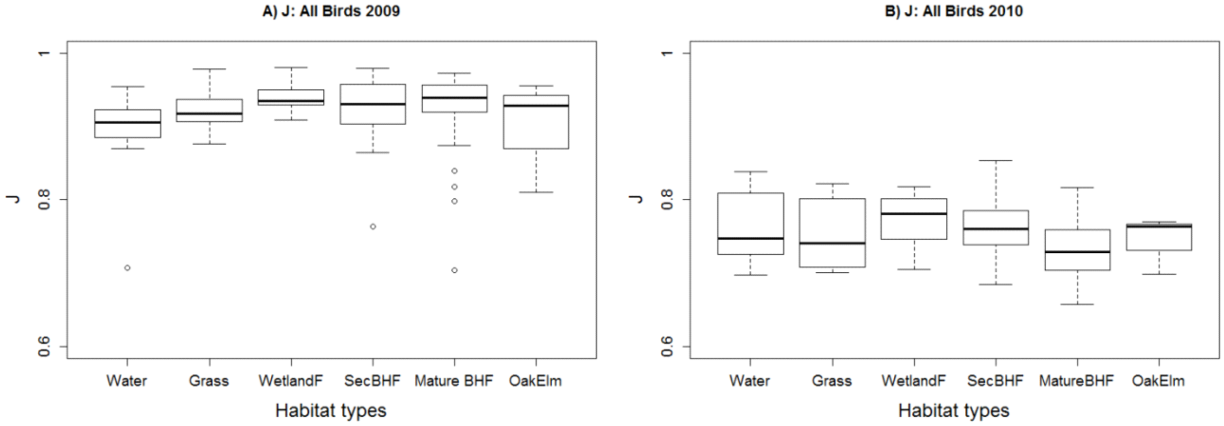


Figure 4.16. Evenness index (J) calculated for all birds across habitat types for A) 2009 and B) 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

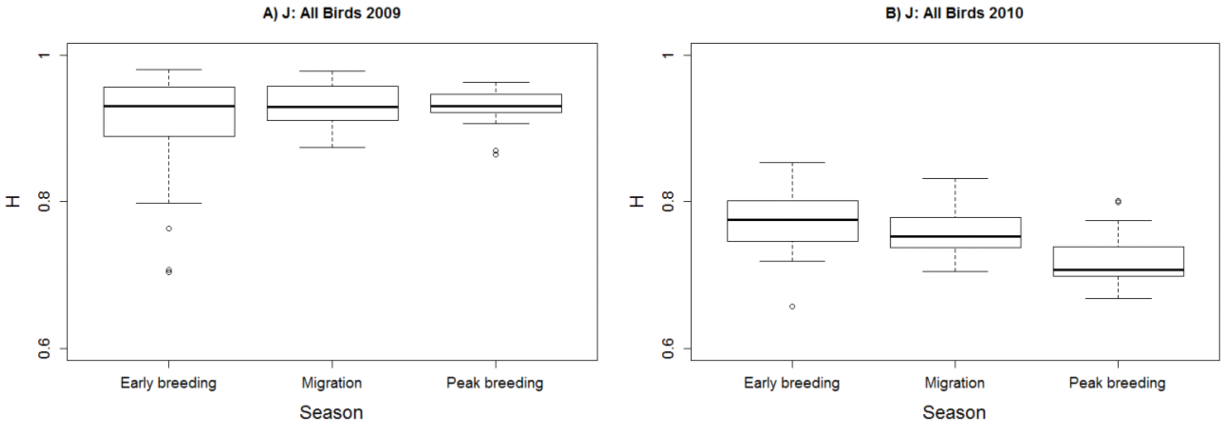


Figure 4.17. Temporal differences in evenness index (J) calculated for all birds in A) 2009 and B) 2010.

4.5.2.2 Local Breeding Birds vs. Non-Local Breeding Birds

Species diversity and richness were higher in the local breeding bird communities than the other group consisting of both wintering species and migratory flyovers. During both census years, mature bottomland hardwood forests had the highest diversity across all habitat types in 2009 ($F_{5,105} = 5.56$, p-value < 0.01) and 2010 ($F_{5,105} = 7.82$, p-value < 0.01) when measured by Shannon-Wiener diversity index (H) (*Figure 4.18*). Mature bottomland hardwood forests had consistently the highest diversity across all habitat types when measured by Simpson's (D) (*Figure 4.20*; $F_{5,105} = 3.26$, p-value < 0.01 for 2009 and $F_{5,105} = 5.79$, p-value < 0.01 for 2010). Further, Tukey's multiple comparison procedure ($\alpha = 0.05$) identified that in 2009, species diversity measured by either index was significantly higher in mature bottomland forests than wetland forests and secondary forests (p-values < 0.01). In 2010, mature bottomland hardwood forests had higher diversity than all habitat types except for oak-elm forests (*Figure 4.18 & Figure 4.23*, p-value < 0.05). Seasonal effect on both diversity indices was significant. In 2009, species diversity measured by (H) and (D) showed a steady and significant increase throughout the sampling season ($F_{2,108} = 17.31$, p-value < 0.01 and $F_{2,108} = 1.13$, p-value = 0.32, respectively) and 2010 ($F_{2,108} = 29.38$, p-value < 0.01 and $F_{2,108} = 19.97$, p-value < 0.01, respectively). See Figure 4.19 & Figure 4.24.

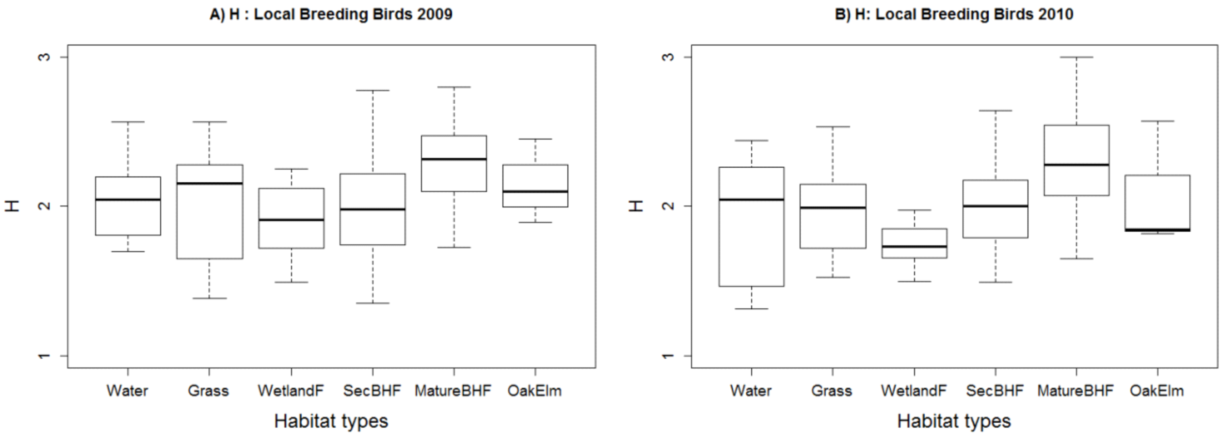


Figure 4.18. Shannon-Wiener diversity index (H) calculated for local breeding birds across habitat types in A) 2009 and B) 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

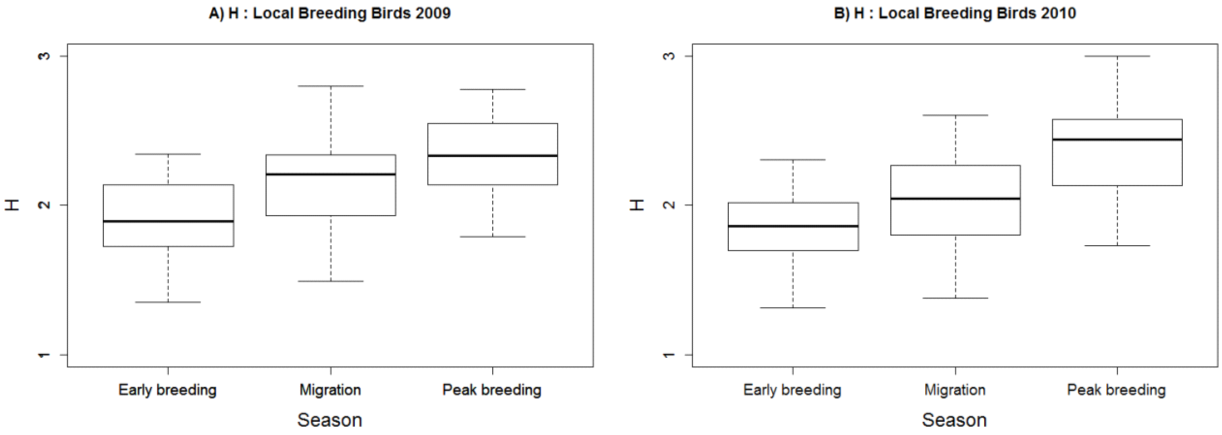


Figure 4.19. Temporal differences in Shannon-Wiener diversity index (H) calculated for local breeding birds across habitat types in A) 2009 and B) 2010.

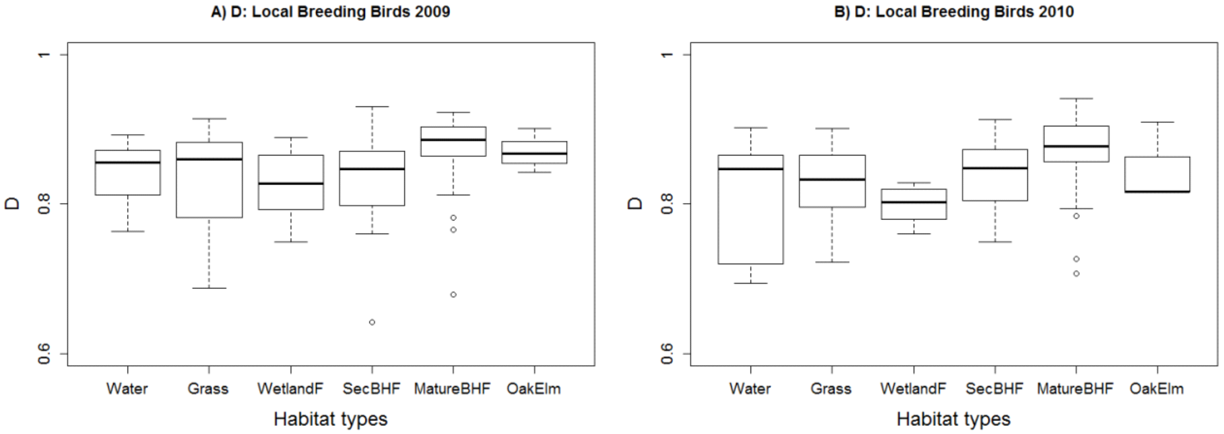


Figure 4.20. Simpson's diversity index (D) calculated for local breeding birds across habitat types in A) 2009 and B) 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

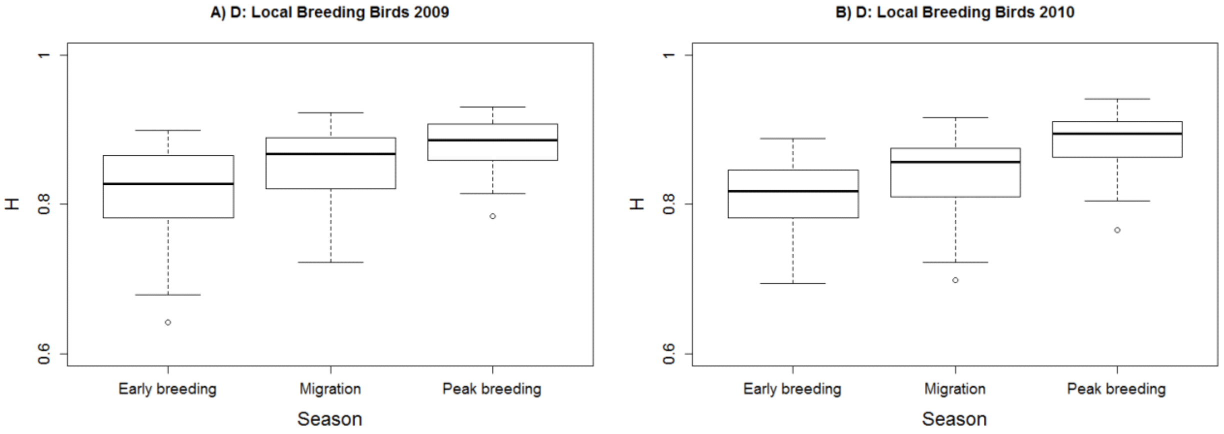


Figure 4.21. Temporal differences in Simpson's diversity index (D) calculated for local breeding birds across habitat types in A) 2009 and B) 2010.

For local breeding bird communities, evenness (*J*) was unaffected by neither by habitat types (*Figure 4.22*; $F_{5,105}$ = 0.85, p-value = 0.51 for 2009 and $F_{5,105}$ = 0.52, p-value = 0.75 for 2010) or sampling season (*Figure 4.23*). Despite the departure of wintering species and arrival of neo-tropical migrants, local breeding bird communities retained relatively consistent evenness throughout the sampling season in both census years ($F_{2,108}$ = 1.13, p-value = 0.32 for 2009 and $F_{2,108}$ = 1.68, p-value = 0.19 for 2010).

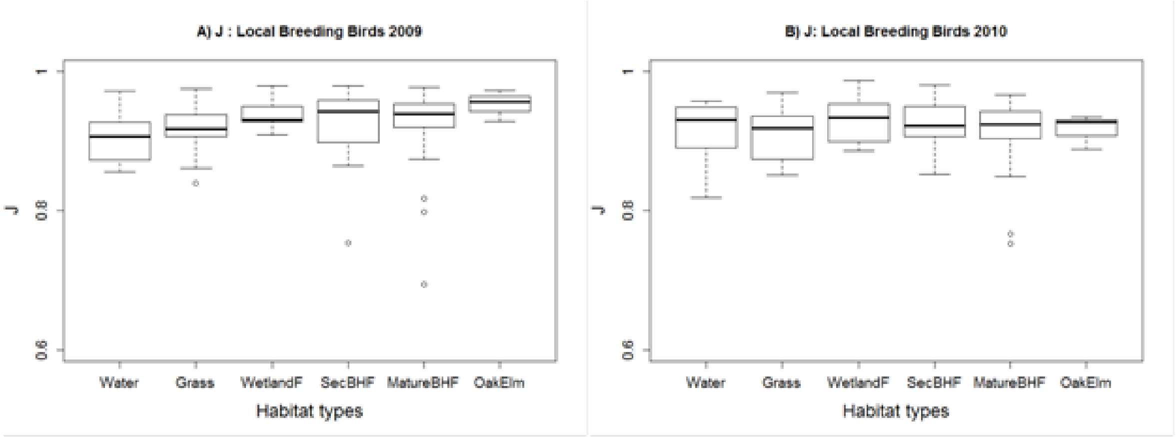


Figure 4.22. Evenness (J) calculated for local breeding birds across habitat types in A) 2009 and B) 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

Figure 4.23. Temporal differences in evenness (J) for local breeding birds observed in A) 2009 and B) 2010.

As for the other group of wintering and migratory flyovers, species diversity measured by Shannon-Wiener index (*H*) was affected only by habitat types (*Figure 4.24*); $F_{5,105} = 4.27$, p-value < 0.01 for 2009 and $F_{5,105} = 2.82$, p-value =0.01). Tukey's multiple comparison tests identified the highest diversity in water sites in both census years (*Figure 4.24*). Seasonal difference was not detected in 2009 ($F_{2, 108} = 1.91$, p-value = 0.15) and in 2010 ($F_{2,108} = 1.56$, p-value =0.21). On the other hand, Simpson's diversity (*D*) which reflects dominance (to abundant species than rare species) produced a different pattern. In 2009, species diversity was unaffected by habitat types ($F_{5,105} = 1.28$, p-value = 0.27), but influenced by the sampling season ($F_{2,108} = 3.58$, p-value = 0.03), with the highest diversity (*D*) observed during migration period. This indicates that the probability of observing rare species varied from place to place, regardless of habitat types, yet was affected by sampling season. In 2010, diversity (*D*) was unaffected by neither habitat types ($F_{5,105} = 1.08$, p-value = 0.37) nor by season ($F_{2,108} = 1.13$, p-value = 0.32).

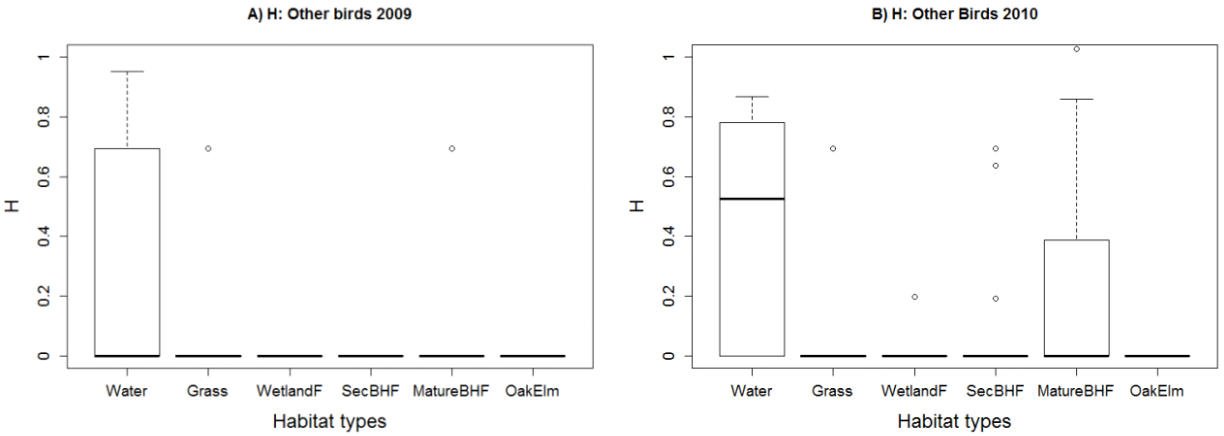


Figure 4.24. Shannon-Wiener diversity index (H) calculated for non-local breeding birds across habitat types in A) 2009 and B) 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

Evenness (*J*) was slightly influenced by habitat types only. In 2009, water sites where the highest diversity (*H*) was detected had the highest evenness (*Figure 4.25*); $F_{5,105} = 3.21$, p-value < 0.01), suggesting a variety of winter or migratory flocking birds stayed at the site. Similarly in 2010, the highest evenness and variability were detected in water sites (*Figure 4.25*); $F_{5,105} = 2.01$, p-value = 0.08). Although insignificant ($F_{2, 108} = 1.96$, p-value = 0.14 for 2009 and $F_{2,108} = 1.20$, p-value = 0.30 for 2010), slightly higher evenness during early breeding season and migration period than peak breeding season.

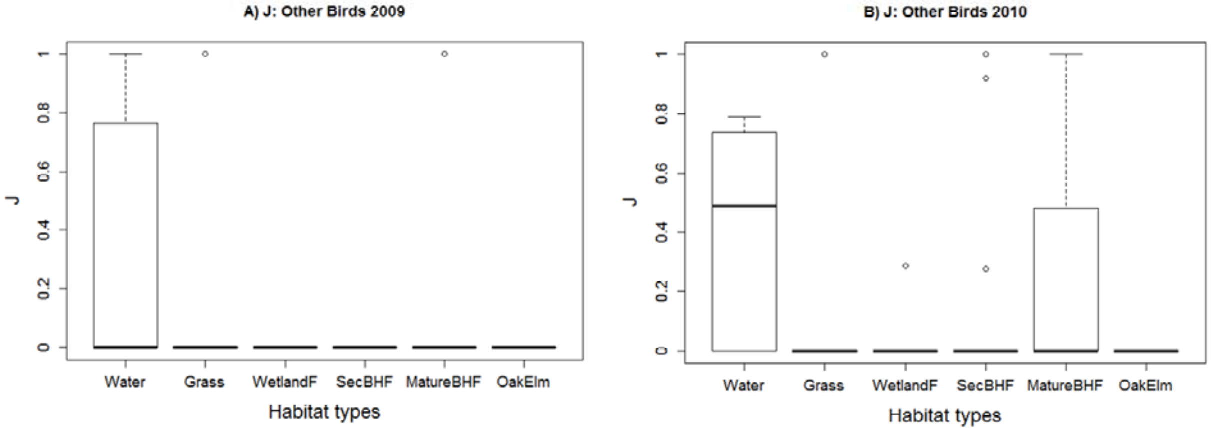


Figure 4.25. Evenness (J) calculated for non-local breeding birds across habitat types in A) 2009 and B) 2010 across six habitat types: water, grass-herb (Grass), wetland forest, secondary bottomland hardwood forest (Sec BHF), mature bottomland hardwood forest (Mature BHF), and oak-elm forest (Oak-Elm).

Overall, community indices calculated for local breeding bird communities mostly coincide with all birds combined. On the other hand, the other group consisting of both wintering birds and migratory flyovers was relatively unaffected by habitat types or sampling season. Both species diversity indices calculated for all birds and local breeding birds were significantly affected by habitat types and sampling season, with an exception of seasonal effect on 2010 for all birds (Table 4.7). Same diversity indices calculated for the other group of birds, consisting of wintering birds and flocking birds, however, show that they were rather independent of habitat or seasonal effects.

The difference between two diversity indices was most noticeable for non-local breeding birds group. By definition, Shannon-Wiener diversity represents the uncertainty of the community by variability of the species composition. Simpson's diversity, in contrast, reflects dominance because it weights the most abundant species more heavily than rare species. It can be inferred that the probability of observing non-local breeding birds (wintering and migratory flyovers) were chance events, thus both habitat and seasonal effects were down weighted for

Simpson's diversity index (*D*). Nonetheless, habitat types had significant effects on species diversity measured by Shannon-Wiener index (*H*) for all bird groups.

Additionally, when looking at both habitat types and sampling timing factors together, while controlling for the other factor, neither habitat or sampling season had significant effect on species diversity measured both by Shannon-Wiener index (*H*) or Simpson's index (*D*), and evenness (*J*) as summarized in Table 4.8 (Two-way ANOVA, all p-values > 0.05). The same pattern held true for all birds included, local breeding birds, and non-local breeding birds group consisting with wintering and migratory flyovers.

Table 4.8

Summary of p-values generated from two-way ANOVA.

Index	Year	All birds	Local breeding birds	Non-local breeding birds
H	2009	0.6	0.39	0.07
	2010	0.83	0.55	0.22
D	2009	0.54	0.55	0.75
	2010	0.47	0.72	0.26
J	2009	0.74	0.87	0.13
	2010	0.19	0.72	0.33

4.5.3 Spatial Distribution Pattern of Local Breeding Birds

4.5.3.1 Ripley's *K* Analysis Results

Multi-distance spatial cluster analysis (Ripley's *K*) was performed to examine spatial distribution pattern of local breeding birds observed within 37 100-m radius circles using ESRI ArcGIS10. *Figure 4.26* shows distribution of local breeding birds observed during three sampling phases. Spatial distribution pattern was determined by the visual interpretation of the Ripley’s *K* plots. Theoretically Ripley’s *K* analysis is highly sensitive to edge effects, which can seriously affect distance-based statistics. To minimize edge effects, points that are located within a distance equal to the maximum distance of an edge of the study area are mirrored so that edge points will have more accurate neighbor estimates. An example for local breeding birds observed in a forest site illustrates a clustering pattern (*Figure 4.27*). Observed spatial pattern (observed *K* values) exceeded expected *K* values (expected random spatial pattern), indicating a clustering pattern. Further, observed spatial pattern exceeded confidence envelopes below distance class of 25 m, suggesting a statistically significant clustering pattern. The appendices summarize the spatial distribution of local breeding birds and associated tree canopies taller than 10 m in height (see Appendices D and E).

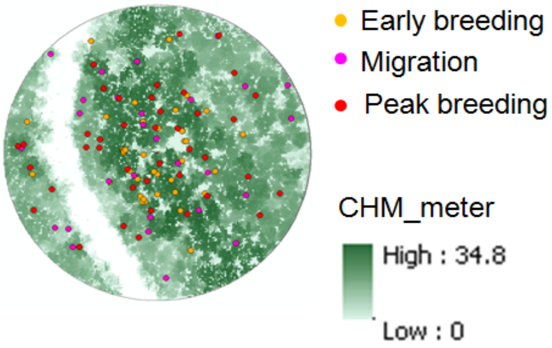


Figure 4.26. Location of observed birds and associated forest canopy height derived from LiDAR.

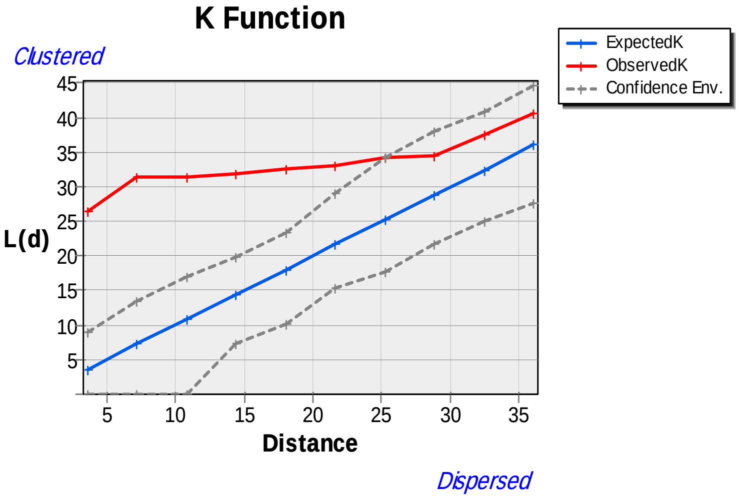


Figure 4.27. Ripley's K calculated for local breeding birds observed in a forest site, showing departures of the observed with respect to expected.

Local breeding birds' spatial distribution patterns within 37 100-m radius circles for each sampling season of 2009 and 2010 are summarized in Table 4.9 based on the percent of sites exhibiting listed patterns.

Table 4.9

Spatial distribution patterns of all birds observed in 2009 and 2010. For each sampling season, percent was calculated by dividing number of sites with birds showing specific distribution patterns by the total number of sites.

	2009			2010		
Season	Early breeding	Migration	Peak breeding	Early breeding	Migration	Peak breeding
Random	33.3%	19.4%	38.9%	38.9%	33.3%	38.9%
Dispersed	0.0%	13.9%	19.4%	8.3%	11.1%	13.9%
Clustered	66.7%	66.7%	41.7%	52.8%	55.6%	47.2%

In both census years, increasing number of sites were occupied by birds which were uniformly distributed (over-dispersed) as season progressed, suggesting a possibility of intensified territorial defense behaviors. This gradual increase in percent of sites occupied by birds with dispersing pattern coincides with the decrease in percent of sites occupied by clustering birds, which may also reflect the possible flocking behaviors occurring in earlier breeding season. When comparing percent of sites with randomly distributed birds vs. non-random (dispersing and clustering patterns) birds throughout the season, birds exhibited the most non-random pattern during migration period in both census years (80.6 % and 66.7 % for 2009 and 2010, respectively). This increase in non-random pattern could be caused either by temporal expansion of local bird population size by migratory flyovers, leading intensified competition against resources, or simply by the habitat preference of migratory flyovers. If, for example, migratory flyovers rest at particular forest stands with specific height characteristics or riparian area, and if those preferred habitat areas had any underlying spatial distribution, birds' habitat preference could simultaneously increase in their spatial distribution pattern.

The increase in non-random bird distribution patterns was affected by habitat vertical heterogeneity associated with the site. Bird spatial distribution pattern, however, seemed irrelevant to the percent of forest cover (area of forest patches taller than 10 m in height occupying the 100 m radius circle at each site). Sites were classified to two groups (homogeneously structured vs. heterogeneously structured sites) by vegetation surface vertical heterogeneity. Vertical complicity of stand structure was determined by standard deviation of canopy surface elevation. Sites associated with relatively larger standard deviations (greater than five) was considered as heterogeneous sites, while sites associated with smaller standard deviation (two to five) was considered as homogeneous sites. An assumption made for this classification was that mature forests where gaps are formed will have a greater vertical heterogeneity in height as compared to younger transitional forests in which stand structure is presumed to be more uniform.

Ripley's *K* analysis results for birds observed in homogeneously structured sites vs. heterogeneously structured sites for each sampling season of 2009 and 2010 are summarized in Table 4.10 and Table 4.11, respectively. In 2009, homogeneously structured sites located in open-field or younger transitional forests were occupied mainly by birds forming no spatial pattern (random) and the ratio of randomly distributed birds gradually increased as the season progressed (35.3 %, 41.2 %, and 58.8 % in early breeding season, migration, and peak breeding season, respectively). Random distribution pattern was gradually replaced by dispersed pattern which potentially reflects intensified territorial behaviors as season progressed. The ratio of clustered pattern gradually decreased throughout the sampling season.

Table 4.10

Spatial distribution patterns of local breeding birds observed in 2009 for two different vertical structure as given by standard deviation (SD) of vegetation height in m. Percent of sites with birds showing specific distribution patterns are summarized for each season.

2009						
Season	Early breeding	Migration	Peak breeding	Early breeding	Migration	Peak breeding
SD tree height	2-5 (vertically homogeneous sites)			> 5(vertically heterogeneous sites)		
Random	35.3%	41.2%	58.8%	31.6%	0.0%	21.1%
Dispersed	0.0%	17.6%	5.9%	0.0%	10.5%	31.6%
Clustered	64.7%	41.2%	35.3%	68.4%	89.5%	47.4%

Table 4.11

Spatial distribution patterns of local breeding birds observed in 2010 for two different vertical structure as given by standard deviation (SD) of vegetation height in m. Percent of sites with birds showing specific distribution patterns are summarized for each season.

2010						
Season	Early breeding	Migration	Peak breeding	Early breeding	Migration	Peak breeding
SD tree height	2- 5 (vertically homogeneous sites)			> 5(vertically heterogeneous sites)		
Random	52.9%	58.8%	70.6%	26.3%	10.5%	10.5%
Dispersed	11.8%	11.8%	11.8%	5.3%	10.5%	15.8%
Clustered	35.3%	29.4%	17.6%	68.4%	78.9%	73.7%

Conversely, in heterogeneously structured forests located in mature forests, higher numbers of sites show randomly distributed birds in early breeding season (31.6 %) than migration (0 %) and peak breeding season (21.1 %). Birds showed an increasing trend of territorial behaviors reflected in a dispersed distribution pattern (0 % in early breeding season, 10.5 % in migration, and 31.6 % in peak breeding season). Clustering pattern which is presumably caused by winter flocking behaviors decreased over time. Similar declining seasonal trends occurred in 2010 as well, with an exception of a consistent dispersed pattern in homogeneously structured sites. More homogeneously structured sites were occupied by randomly distributed birds as season progressed. This ratio was reversed in heterogeneously structured forests and more sites become occupied by birds with dispersed pattern, indicating potential intensified territorial behaviors.

While additional field studies are required, Ripley's *K* analysis results indicate that site specific spatial distribution of local breeding birds could be influenced by sampling season, potentially reflecting winter flocking behaviors or social aggregation. Results also suggest that the intensity or extent of bird spatial distribution pattern is highly influenced by habitat vertical heterogeneity measured by standard deviations of vegetation surface height.

4.5.3.2 Importance of Direction

Relative position (score) of each bird species in ordination space with respect to environmental gradients measured by x-y coordinates for 2009 and 2010 local breeding bird communities are shown in *Figure 4.28* and *Figure 4.32*.

To further assess any potential directional concentration trend in local breeding bird spatial distribution, a directional semivariogram was calculated for each species. Directional responses varied among species. For example, barred owl (*Strix varia*) semivariance didn't show increase to a sill for any direction (*Figure 4.30*), indicating random distribution across the study site. In contrast, painted bunting (*Passerina ciris*) had a semivariance with bounded increase running 0° (North-South) and 45° (Northeast - Southwest) gradients across the sample space, yet decreasing semivariance along 90° (East-West) and unbounded semivariance along 135° (Northwest-Southeast) (*Figure 4.31*). Similarly, mourning dove (*Zenaida macroura*) semivariance resulted unbounded along 135° (Northwest-Southeast), bounded along 0° (North-South) and 45° (Northeast - Southwest), and decreasing along 90° (*Figure 4.32*).

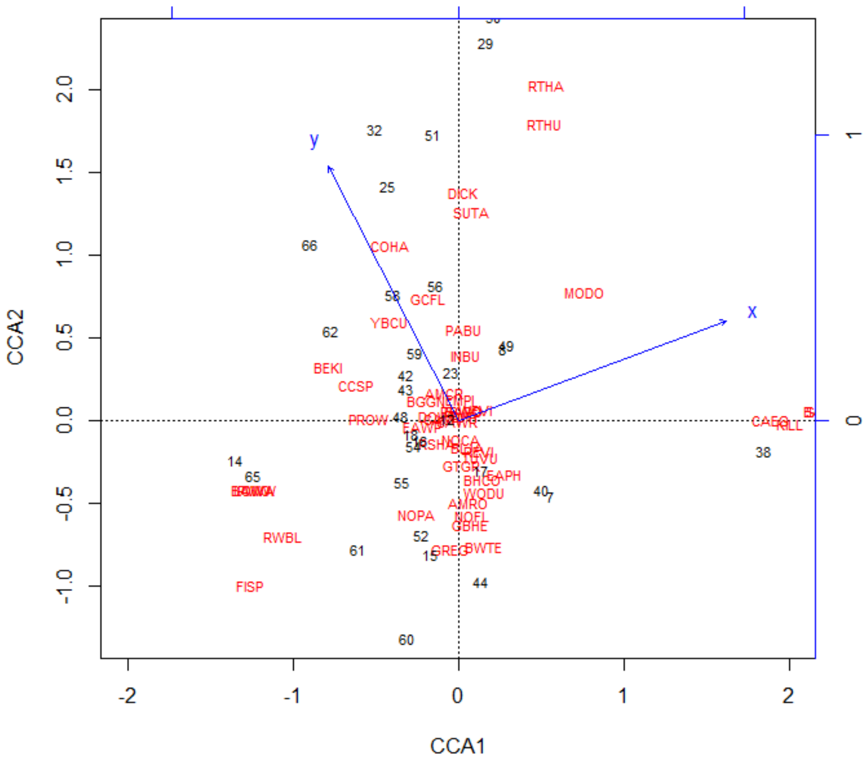


Figure 4.28. A CCA plot showing the effect of x-y coordinates on the 2009 bird communities.

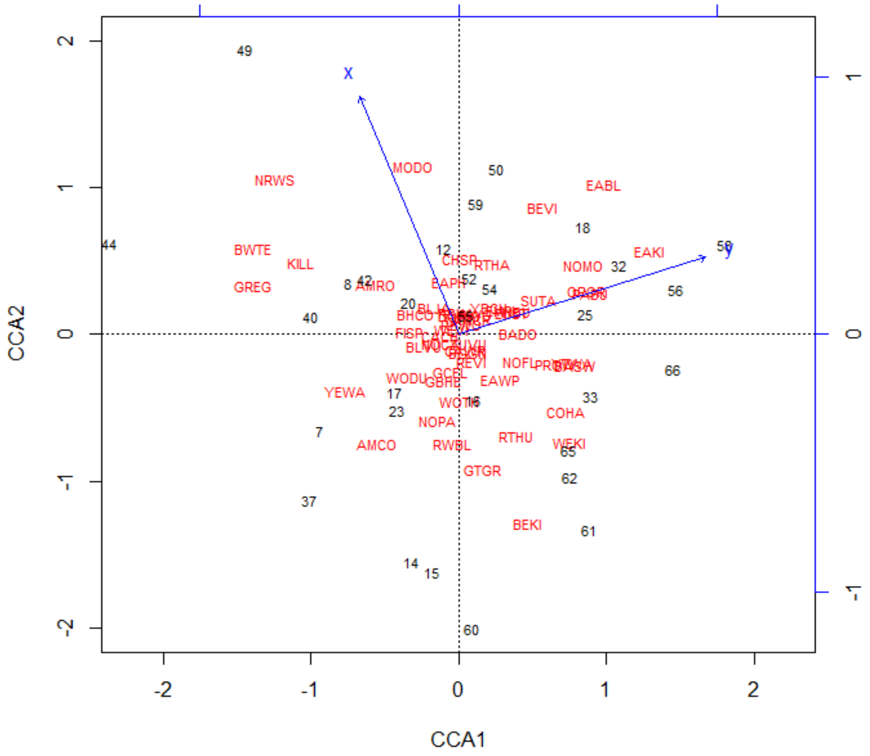


Figure 4.29. A CCA plot showing the effect of x-y coordinates on the 2009 bird communities.

Anisotropic distribution patterns of birds such as painted bunting and mourning dove could be caused by heterogeneous distribution of preferable environmental conditions for each species. Visual estimates of approximate parameter values of variogram models for each species detected in 2009 and 2010 are summarized in Table G.5.1 and Table G.2, respectively.

4.5.3.3 Directional Variograms

Table 4.12 lists species that exhibited bounded semivariances along 0° (North-South). These species include forest specialist foliage gleaners such as indigo bunting, red-eyed vireo, and summer tanager (*Piranga rubra*), forest flycatchers, and some other open woodland ground foragers such as carolina wren (*Thryothorus ludovicianus*) and painted bunting. Similarly, bark foragers such as red-bellied woodpecker (*Melanerpes carolinus*) and downy woodpecker (*Picoides pubescens*) show higher sill along 0° (North-South) of the study site. Anisotropic distribution pattern of these species may reflect the availability of suitable habitat space, described by the extent and connectivity of mature forest patches. It is clear that the riparian forests running from North-South of the study site plays an important role in facilitating the movement of those species and serving as a functional corridor.

In contrast, habitat generalists, such as omnivorous american crow (*Corvus brachyrhynchos*) and blue jay (*Cyanocitta cristata*), large-bodied raptors which include turkey vulture (*Cathartes aura*) and barred owl, and species found in open woodland or shrubby habitat, such as white-eyed vireo (*Vireo griseus*) and northern cardinal (*Cardinalis cardinalis*), and omnivorous carolina chickadee (*Poecile carolinensis*) exhibited random spatial distributions for both census years (see APPENDIX G).

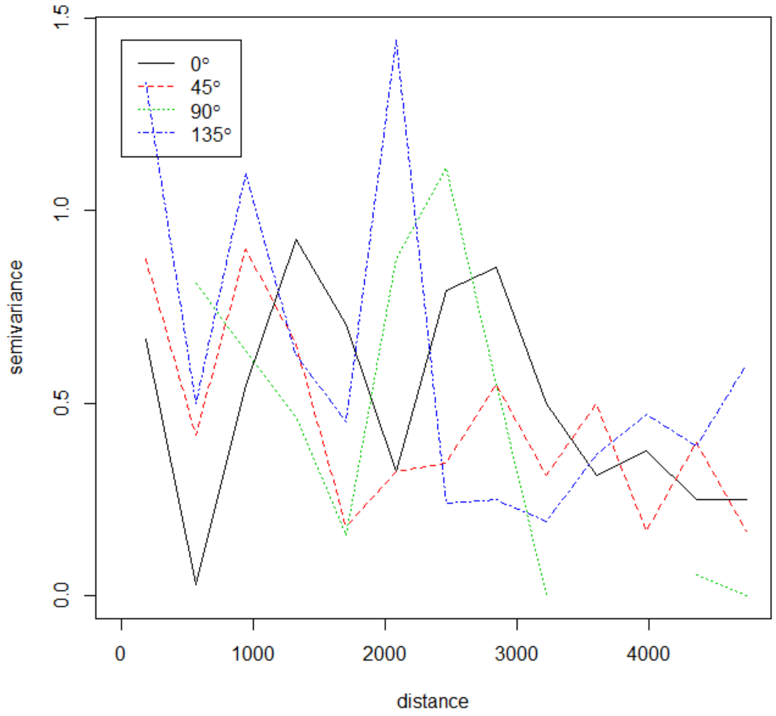


Figure 4.30. Variograms of the observed barred owl pattern based on the 37 locations in the study site. Four directional semivariograms (0°, 45°, 90° and 135°) were calculated with a lag interval of 500 m and a maximum lag distance of 5000 m.

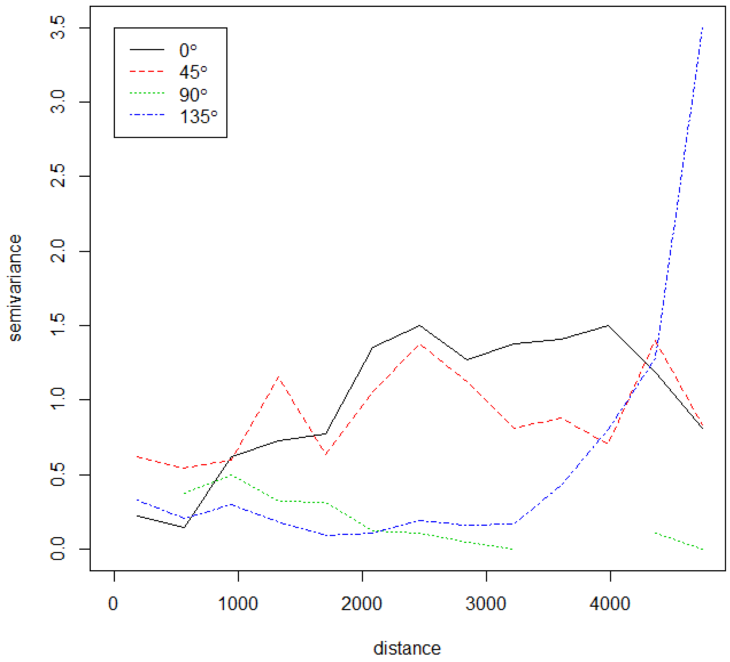


Figure 4.31. Variogram of the observed painted bunting pattern based on the 37 locations in the study site. Four directional semivariogram (0°, 45°, 90° and 135°) were calculated with a lag interval of 500 m and a maximum lag distance of 5000 m.

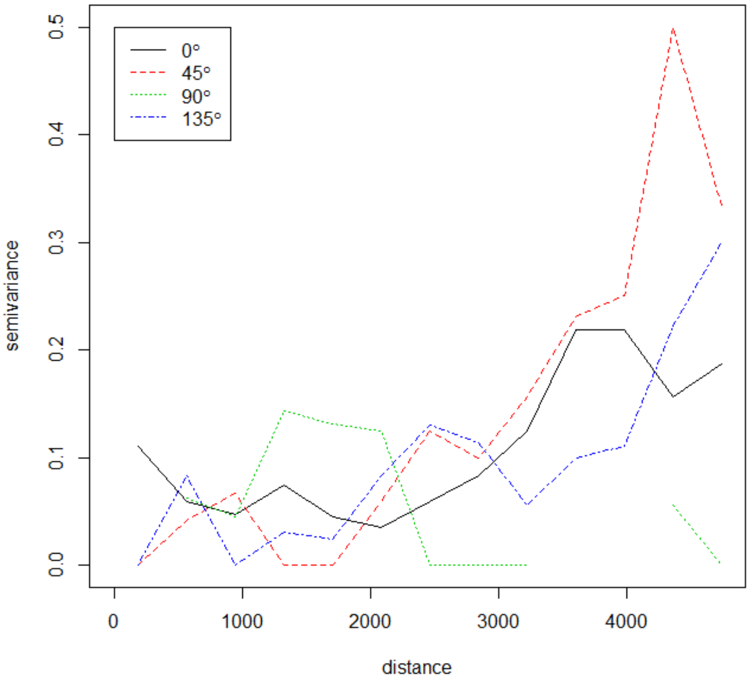


Figure 4.32. *Variograms of the observed mourning dove pattern based on the 37 locations in the study site. Four directional semivariograms (0°, 45°, 90° and 135°) were calculated with a lag interval of 500 m and a maximum lag distance of 5000 m.*

Table 4.12

Species that showed sills along North-South variogram analysis for local breeding birds detected at 37 census points during breeding seasons of 2009 and 2010.

	Species	Year
Woodpeckers	Red-bellied Woodpecker	2009, 2010
	Downy Woodpecker	2009, 2010
Foliage gleaners	Indigo Bunting	2009, 2010
	Red-eyed Vireo	2009, 2010
	Summer Tanager	2009, 2010
	Blue-gray Gnatcatcher	2009, 2010
	Yellow-billed Cuckoo	2009, 2010
	Tufted Titmouse	2009, 2010
	Orchard Oriole	2010
	Bell's vireo	2010
Flycatchers	Great Crested Flycatcher	2009, 2010
	Eastern Wood-Pewee	2009, 2010
Forest species	Carolina Wren	2009, 2010
	Painted Bunting	2009, 2010

4.5.4 Bird Community Structure across Habitat Types

Semivariogram analysis is informative to investigate each bird species' directional distribution pattern across the study site, yet it doesn't offer statistical significance nor take into account coexisting species that could influence the spatial distribution of birds' abundance and distribution. To examine differences in bird community structure across habitat types and visualize distance among species or sites, non-metric multi-dimensional scaling (NMDS) was employed. To analyze ecological distance (differences in species composition among sites), Bray-Curtis dissimilarity index was calculated and results were plotted in ordination space.

Bird community composition showed a well-defined position across habitat types indicated by different colors separated by ellipses for 2009 (*Figure 4.33*) and 2010 (*Figure 4.34*). Ordination methods geometrically arrange sites so that distances among sites in the graph represent their ecological distances. Sites that are close together in the ordination space are interpreted as being similar in species composition, whereas sites separated away interpreted as containing different species composition. Both in 2009 and 2010, bird species observed in mature bottomland hardwood forests were closer to (and partially overlapped with) species observed in secondary forests, indicating a high similarity in bird species composition between these habitat types. Variability in bird assemblages in secondary forests was larger (indicated by slightly larger ellipses), exhibiting differences in species composition which reflect differences in habitat condition associated with different age cohorts. Variability observed in water sites in 2010 (*Figure 4.34*) was slightly larger than the one in 2009 (*Figure 4.33*) on NMDS plots and this pattern was consistent with ANOVA plots for both census years (*Figure 4.12*).

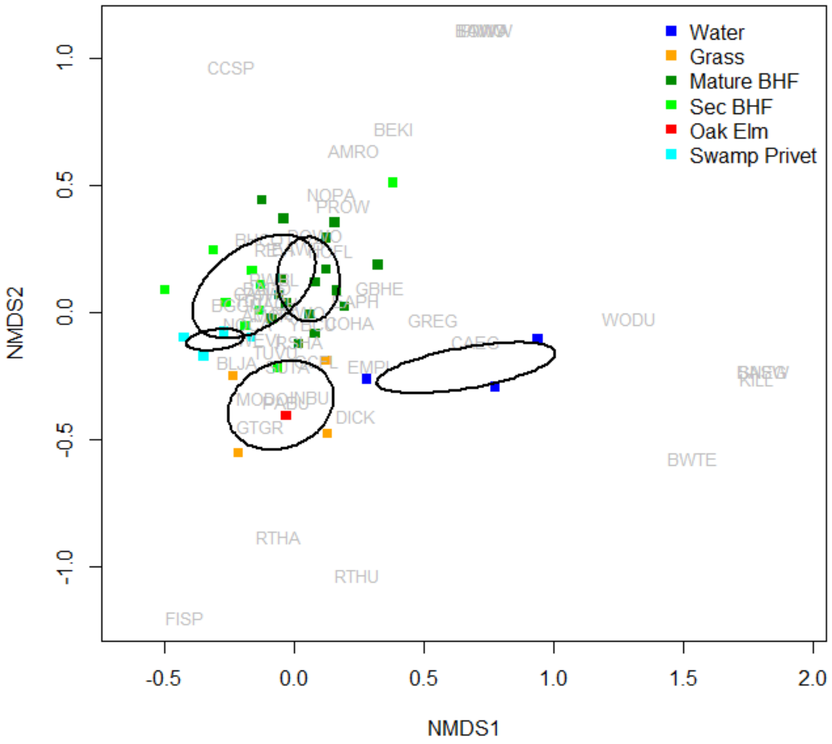


Figure 4.33. NMDS plots showing differences in 2009 bird community composition across habitat types.

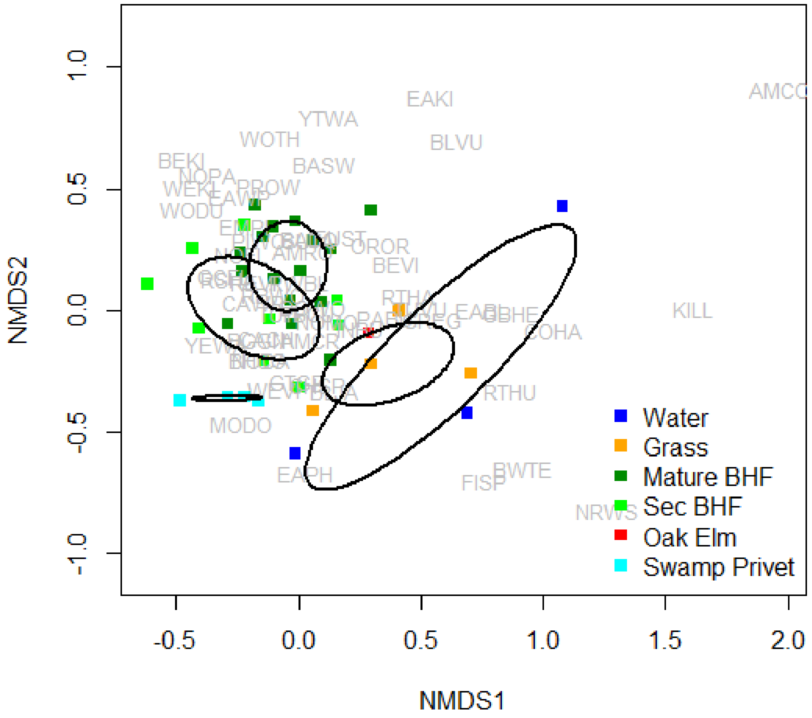


Figure 4.34. NMDS plots showing differences in 2010 bird community composition across habitat types.

Although no conclusive results can be drawn without further data sampling, there is a possibility that weather conditions contributed to the local food availability, resulting in the difference in the amount of variability observed at water sites. Total precipitation, monthly average temperature, average of monthly maximum and minimum temperature, and standard deviations of each temperature parameter during the breeding season (roughly February to May) are summarized in Table 4.13. Temperature and precipitation data was measured at the weather station in the GBC for 2009 and the University of North Texas Discovery Park (located approximately six miles west of the GBC) for 2010. In North Texas, greater precipitations, higher temperatures, and stability (indicated by lower standard deviations) in spring generally indicate more preferable conditions to maintain larger insect populations. Weather data suggests that 2009 might have been a better year for birds as compared to 2010, which had lower precipitation and larger fluctuations in temperatures.

Table 4.13

Summary of weather conditions during breeding seasons of 2009 and 2010.

	2009	2010
Total precipitation(cm)	34.1	9.9
Monthly average temperature (C^0)	16.6	16.8
Average of monthy maximum temperature (C^0)	32.2	27.5
Average of monthy minimum temperature (C^0)	-2.3	0.8
Standard deviation of average temperature (C^0)	7.3	7.8
Standard deviation of monthy maximum temperature (C^0)	1.6	6.9
Standard deviation of monthy minimum temperature (C^0)	2.1	5.1

Despite a slight difference in the relative position of each habitat types represented by bird community composition, however, bird communities observed in 2009 and 2010 showed similar distribution patterns across habitat types. Forest bird communities were generally closer together and separated away from other habitat types such as open fields or water sites. Wetland forests had distinctive bird communities located between the remaining two clusters: forested sites and relatively open sites (grassland-herbaceous sites and water sites).

The amount of stress calculated for NMDS analysis was 0.22 and 0.21 for 2009 and 2010, respectively. Relatively high stress indicates weak ties or poor goodness of fit (Kruskal, 1964). Previous research has argued that the amount of stress only gives a vague indication of the goodness of fit and could be influenced by the dimensionality of the data. Because stress decreases as the number of dimensions increases, a two-dimensional NMDS always has more stress than a three-dimensional one. The Shepard diagram is a technique commonly used for judging the adequacy of a NMDS (Hair et al., 1998). Shepard function in R-vegan package plots ordination distances against original dissimilarities, and draws a step line of the nonlinear fit. In addition, it adds to the graph two correlation-like statistics on the goodness of fit. The nonmetric fit is based on stress (*S*) and defined as $\sqrt{(1 - s^2)}$. The "linear fit" is the correlation between fitted values and ordination distances (Oksanen, 2013). Shepard diagrams produces to measure the relationship between the proximities (observed dissimilarities) and the distance of the point configuration for 2009 and 2010 are displayed in Figure 4.35 and Figure 4.39. Non metric fit was 0.95 and linear fit was approximately 0.8 for both census years, indicating a good fit (Hair et al., 1998).

Additionally, analysis of similarities (ANOSIM) was applied to test if the assigned habitat group was meaningful. If birds observed at each habitat types were meaningful, bird community composition within habitat types should be more similar in composition than birds from different habitat types. ANOSIM statistic values of 0.71 (2009) and 0.61 (2010) indicate relatively high similarities within groups, which can be translated to significant differences in bird species composition across habitat types for both census years (p-values = 0.001).

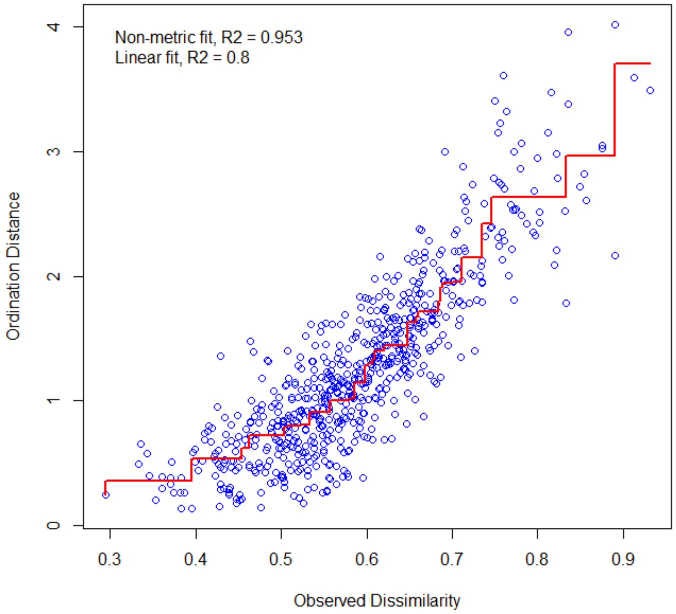


Figure 4.35. Shepard diagrams produces to measure the relationship between the proximities and the distance of the point configuration for 2009.

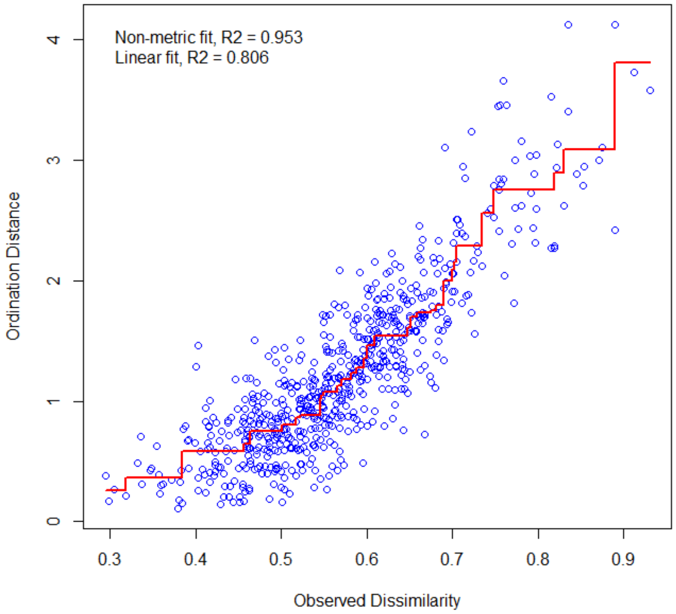


Figure 4.36. Shepard diagrams produces to measure the relationship between the proximities and the distance of the point configuration for 2010.

4.5.5 Environmental Variables Selection

After checking multicollinearity and variance inflation factors, four environmental variables were selected. Largest patch index (LPI) and connectivity (CONNECT) indices were based on forest patch metrics calculated for forest patches taller than 10 m in height. LPI and CONNECT were log transformed to better meet normality assumption. Coefficient of variation (CV) of tree height measures vertical heterogeneity of vegetation surface. Habitat types measure differences in vegetation communities.

In 2009 and 2010, 36.02 % and 42.12% of variation in bird abundance was explained by selected environmental variables, respectively. Vertical heterogeneity of canopy surface measured by CV was the most influential variable, followed by connectivity of forest patches in both census years. The first two CC axes explained 40.46% in 2009 and 45.05 % in 2010. In 2009, both axes were positively correlated with LPI and negatively correlated with CONNECT and CV of tree height (*Figure 4.37*). Axes 1 broadly separated forested (vegetated) sites from sites that were relatively open and axes 2 seemed to separate sites along vegetation height gradient, yet the correlation is weak. In 2010, axis 1 clearly separated forested sites from open fields and axes 2 separating sites along vegetation height gradients (*Figure 4.38*). These results suggest that each species forage in a distinct space. At the same time, a substantial overlap in preferred foraging space was observed for some species.

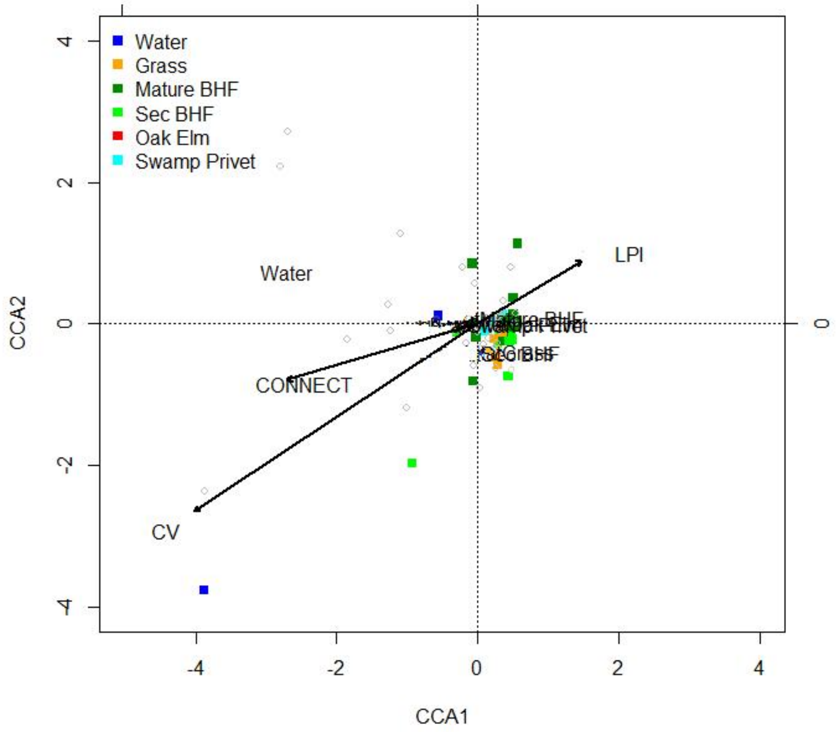


Figure 4.37. A CCA plot showing relative scores on sites and bird species for 2009.

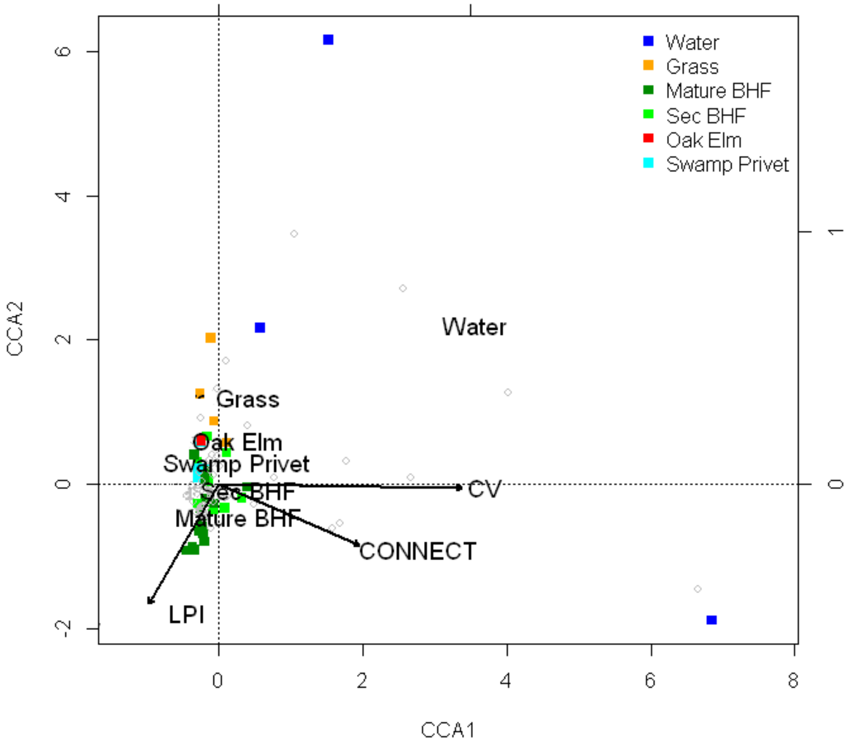


Figure 4.38. A CCA plot showing relative scores on sites and bird species for 2010.

When geographical coordinates are added to the set of environmental variables, 42.64 % and 47.41 % of variation in bird abundance was explained in 2009 (*Figure 4.37*) and 2010 (*Figure 4.38*) (the first two axes accounted for 44.23 % and 45.95 % of the variation).

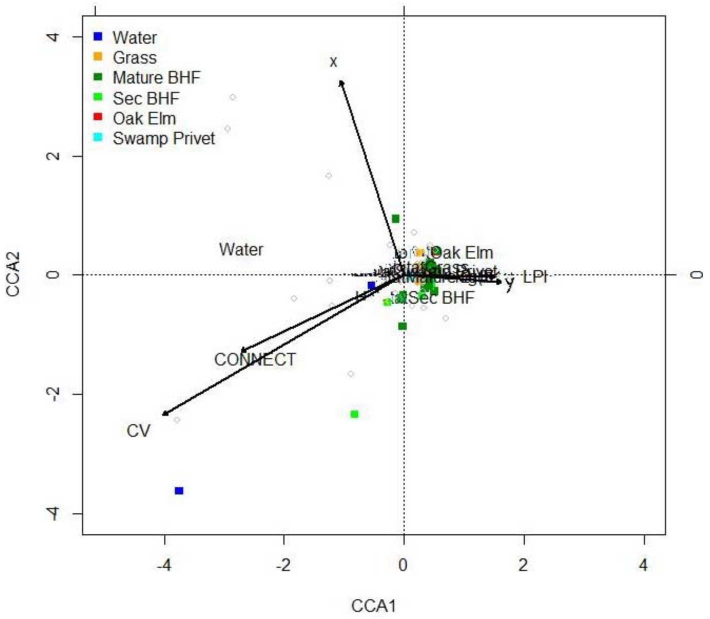


Figure 4.39. A CCA plot showing relative position of sites and bird species and environmental variables affecting differences in the 2009 bird community composition.

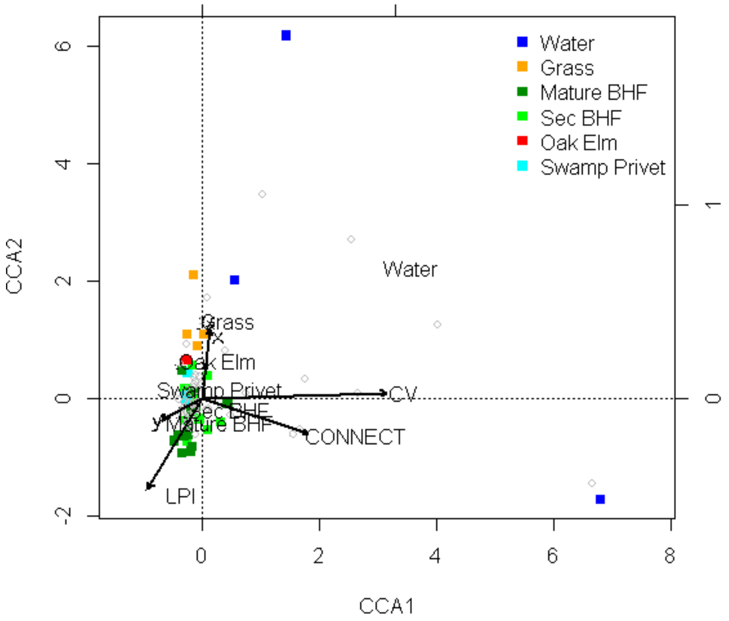


Figure 4.40. A CCA plot showing relative position of sites, bird species, and environmental variables affecting differences in the 2010 bird community composition.

4.5.6 Variation Partitioning

Adding spatial information (x-y coordinates) increased the percent of explained variation in bird community assemblages, however, this doesn't account for the potential autocorrelation of environmental variables or bird-environmental relationships. Ripley's *K* analysis has shown that local breeding bird distribution pattern is highly influenced by forest vertical heterogeneity measured by standard deviation of canopy height surface within 100 m radius circles surrounding the bird census points. Further, at a broader scale, some of the local breeding birds, especially forest canopy species, have shown a directional distribution gradients along riparian forests running North-South of the study site. These results suggest that spatial distribution patterns of local breeding birds is correlated with the distribution pattern of forests at various scales.

To assess the relative contribution of environmental variables and space to the bird community assemblages, partial CCA was performed. Variance partitioning results for 2009 and 2010 are summarized in Figure 4.41 and Figure 4.45, respectively. In 2009, selected environmental variables (coefficient of variation of tree height, connectivity of forest patches taller than 10 m in height, largest patch index, and habitat types) explained 43.48 % of the total variation. Of the total variance explained, 86.55 % was explained by environment alone which was not shared by space, 10.73 % was explained by space alone, and 2.72% was explained by spatially structured environmental variables. Similar results were obtained for 2010. Selected environmental variables explained 48.68 % of the total variance, of which 85.64 % was explained by environment alone which was independent of spatial effects, 13.48 % was explained by space alone, and only 0.88 % was explained by spatially structured environmental variables.

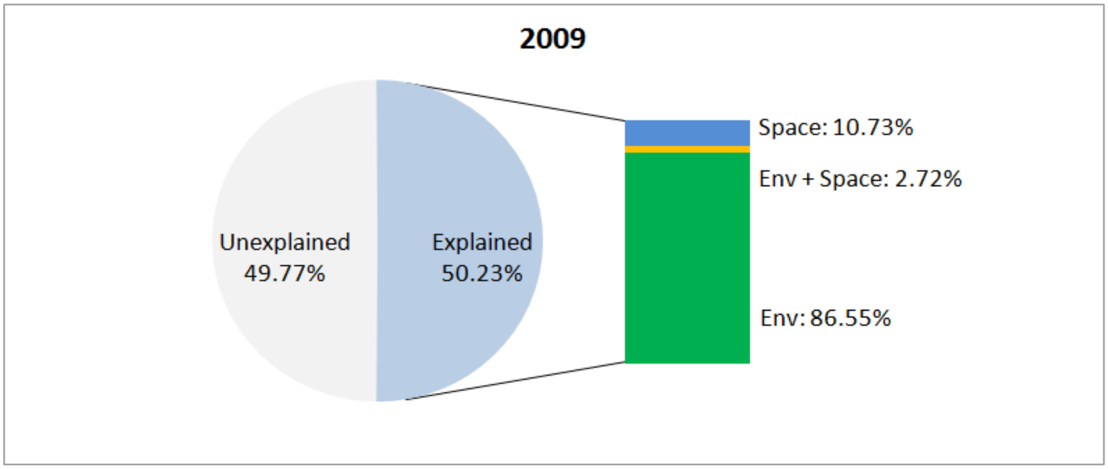


Figure 4.41. Variation partitioning of 2009 local breeding birds assemblages.

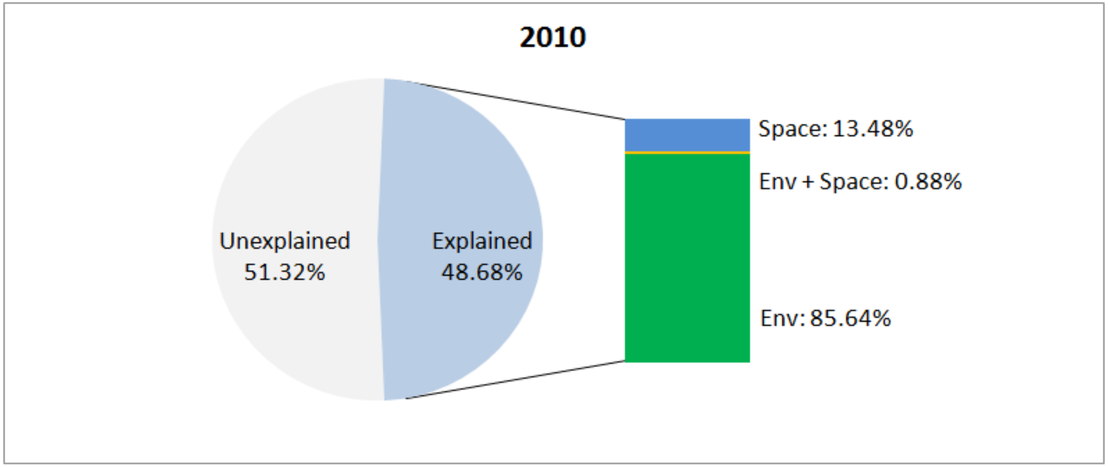


Figure 4.42. Variation partitioning of 2010 local breeding birds assemblages.

The total amount of variation explained by environmental variables, spatially structured environmental variables, space alone (50.22 % for 2009 and 48.68 % for 2010) was consistent with other similar studies with variation ranging from 41 % to 57 % (Borcard & Legendre, 1992; Hobson, 2000; Titeux et al., 2004). Proportion explained by environmental variables alone in this study (43.5 % in 2009 and 41.7 % in 2010), however, turned out much higher than other studies (14 %-27 %). Proportions of spatially structured environmental variation were highly variable among studies ranging from 3 % to 31 %, which can exceed environmental alone variation or space variation (Borcard & Legendre 1998; Hobson et al., 2000; Titeux, 2004). Low fraction of spatially-structured environmental variation observed in this study indicates that bird community composition and selected environmental variables, most of which were based on forest patch metrics taller than 10 m in height) didn't have common spatial distribution pattern or underling ecological processes.

Variance partitioning signifies that environmental variables explained the majority (over 80-90 %) of variation in bird community assemblages and space turned out to be a minor descriptor of bird community composition at the scale of this study. After factoring out the effect of the selected environmental variables, space alone and spatially structured environmental variables contributed a small percentage of the variation. The small percentages of space alone (10.73 % for 2009 and 13.48 % for 2010) reflect autocorrelation of biological processes which was not shared by four environmental variables selected for this analysis. Biological processes that could have contributed to spatial autocorrelation include predation, competition, or social aggregation of wintering birds or non-reproductive individuals. Although minor, detected autocorrelation corresponds to the directional distribution of some of the local breeding forest birds captured by semi-variogram analysis.

Although bird community assemblages were driven by environment alone or environmental variables, approximately 50 % of the variance still remains unexplained. There are many factors that could have affected results. First, although the study site in the GBC consists of various habitat types, serving as a sanctuary for many bird species, the spatial scale may not large enough to capture the spatial structure of ecological gradients that operate at larger scale. Alternatively, land-use history (partial clear-cut) might have eluded the effects of spatially structured environmental variables. Spatial scale of this study might have contributed to relatively high fraction of unexplained variation. Directional semi-variogram analysis has suggested that some species, particularly forest specialists were highly dependent on North-South gradient, overlapping the extent of riparian forests along the Elm Fork of Trinity River. There were, however, some habitat generalists whose spatial distribution was independent of environmental variables, exhibited by the absence of any directional concentration. These species included large-bodied raptors (vultures and owl) and omnivorous species (i.e. American crow, blue jay). Particularly for larger bodied raptors, there remains a possibility that their foraging range exceeded the spatial extent of this study, potentially increasing unexplained variation. Further, habitat occupancy doesn't always correspond to the saturation of suitable habitat patches. The disproportionate use of a suitable habitat relative to its availability, called habitat selection (Jones, 2001), suggests that habitat saturation will be greatly affected by various biotic interactions which determine food availability. Many biotic interactions such as predation and competition could also influence local breeding birds' abundance and spatial distribution. Moreover, selected environmental variables might not be appropriate for this study. Inclusion of some unmeasured environmental variables such as abundance of shrub layers, more precise stand vertical structure, or the size of sub-canopy gap, all of which are correlated to habitat requirements of some bird species detected in the study site could have increase the total variation explained.

4.5.7 Foraging Guilds Analysis

CCA results suggest that each bird species forage in a distinct space. At the same time, a substantial overlap in preferred foraging space was observed for some species. To investigate how birds in the foraging guilds related to selected environmental variables, 65 local breeding bird species were classified in 16 groups based on the birds' foraging behaviors and basic habitat requirements (for instance aquatic, arboreal, and ground) (Table 4.14). Foraging guilds classification was conducted based on species specific foraging behaviors and general habitat descriptions obtained from the Cornell Lab of Ornithology website (Cornell Lab of Ornithology, 2013).

Table 4.14

Foraging guilds classification of local breeding birds. Carnivorous species mainly consume small mammals, fish, or arthropods other than insects.

Groups	code	Species								
Carnivorous (Riparian)	CR	BEKI	GBHE	GREG	SNEG					
Carnivorous (Terrestrial)	CT	RSHA	RTHA	BADO	TUVU	BLVU	COHA			
Insectivorous (Riparian)	IR	WODU	LOWA							
Insectivorous (Ground-Low vegetation)	IG_low	KILL	EUST	CAEG						
Insectivorous (Ground-woodland)	IG_wood	AMRO	NOFL	FISP	WOTH	CAWR	CHSP			
Insectivorous (Arboreal: Low vegetation)	IA_low	RWBL	EABL							
Insectivorous (Arboreal: Fly catching)	IA_fly	EAWP	GCFL	FMPI	EAPH					
Insectivorous (Low vegetation: Fly catching)	IL_fly	STFL	WEKI	EAKI						
Insectivorous (Aerial forager)	I_air	NRWS	BAWS							
Insectivorous (Arboreal: Canopy-sub canopy)	IA_can	BGGN	NOPA	REVI	YBCU	PROW	YTWA	BAWW	CACH	TUTI
Insectivorous (All vegetation levels)	I_all	WEVI	YEWA	INBU	SUTA	BEVI	OROR			
Insectivorous (Bark forager)	I_bark	DOWO	RBWO	PIWO						
Omnivorous	OG	AMCR	BLJA	GTGR	NOMO	COGR				
Nectarivorous (open woodland)	N_wood	RTHU								
Granivorous (Ground)	GG	NOCA	CAWR	BHCO	MODO	CHSP	DICK	PABU		
Granivorous (Ripatrian)	GR	BWTE	AMCO							

Results were consistent with the previous analysis, for which members of the foraging guilds show a well-defined position on NMDS plots. Ecological distance across each foraging guild in terms of habitat preference described by four selected environmental variables show a well-defined position both for 2009 (*Figure 4.43*) and 2010 (*Figure 4.44*). For both census years, riparian carnivorous, riparian granivorous, low vegetation insectivorous, and aerial insectivorous species, all of which are associated with water-open fields appeared separated from the remaining foraging guilds. At the same time, arboreal flycatchers, woodland insectivores, canopy residents foliage gleaners, and woodland ground granivores were clustered together, in forested habitat.

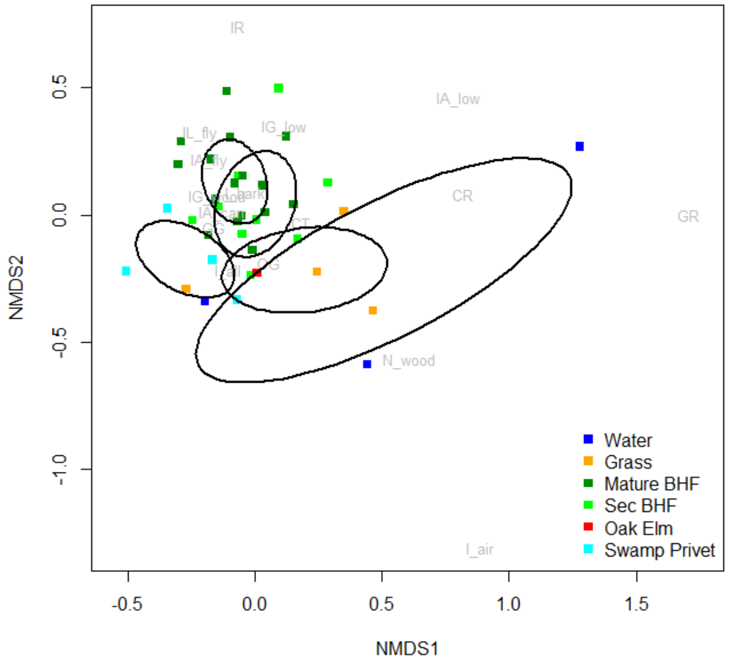


Figure 4.43. A NMDS plot showing differences in the 2010 foraging guilds across habitat types.

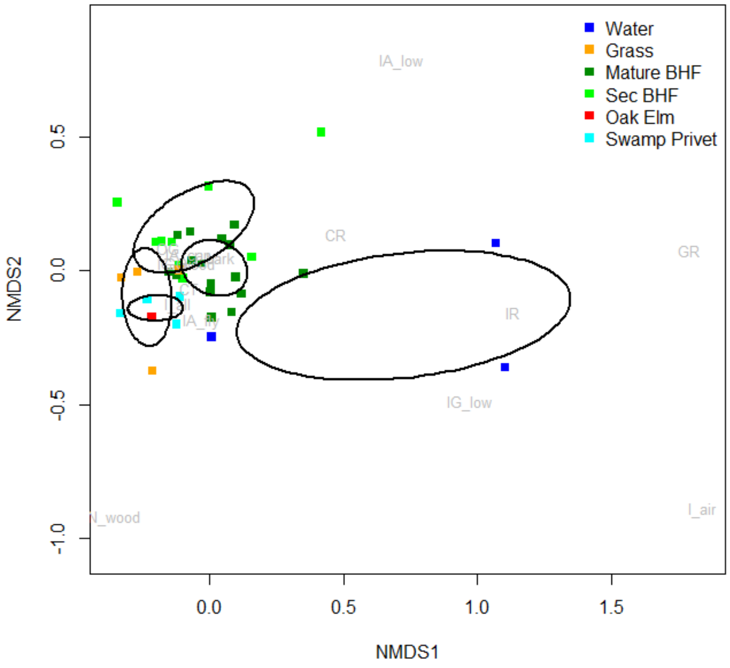


Figure 4.44. A NMDS plot showing differences in the 2010 foraging guilds across habitat types.

Species with more generalist characteristics, such as insectivorous species found in all vegetation layers or omnivorous species, in addition to relatively-large bodied carnivorous raptors were located between woodland species and open fields, suggesting they benefit from forest edges.

CCA analysis on foraging guilds results also demonstrated similar pattern with CCA analysis on individual species. In 2009, selected environmental variables (coefficient of variation of vegetation height surface, connectivity of tree patches taller than 10 m in height, largest forest patch index) explained 50.28 % of total variation in bird foraging guilds, with the first two axes accounted for 36.06%. Similarly in 2010, environmental variables explained 53.32 % of variation in the bird foraging guilds, with the first axes explaining 53.32 % of variation. When compared to CCA results based on species scores across habitat types (42.64 % for 2009 and 47.41 % for 2010), there was a slight increase (7.64 % for 2009 and 5.91 % for 2010) in percent of variation explained by selected variables. Coefficient of variation, which measures the vertical heterogeneity of vegetation surface, had the largest influence among all environmental variables, followed by connectivity of forest patches in 2009 (*Figure 4.45*), whilst in 2010, coefficient of variation, forest patch connectivity, and largest forest patch index had similar effects on the bird foraging groups across habitat types(*Figure 4.46*). Although sites appear more clumped in the ordination space in 2009, results from both census years had similar patterns. Particularly, forest residents which included arboreal insectivores in canopy-sub-canopy layers, arboreal flycatchers, insectivorous bark foragers, insectivores foraging on woodland floors, were associated with mature bottomland hardwood forest patches. As consistent with trends observed in NMDS plots, more habitat generalists such as omnivorous ground forager, insectivorous foraging in all vegetation layers regardless of the vegetation height, and granivorous ground foragers were associated stronger with secondary forests. Additionally, aerial insectivores were separated away from other foraging guilds.

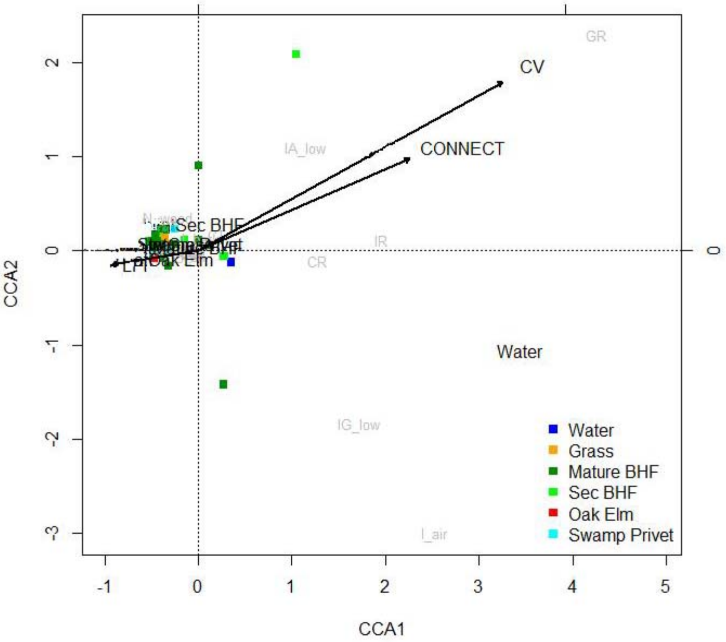


Figure 4.45. CCA plots showing relative position of sites, birds foraging guilds, and environmental variables for the 2009 bird community composition.

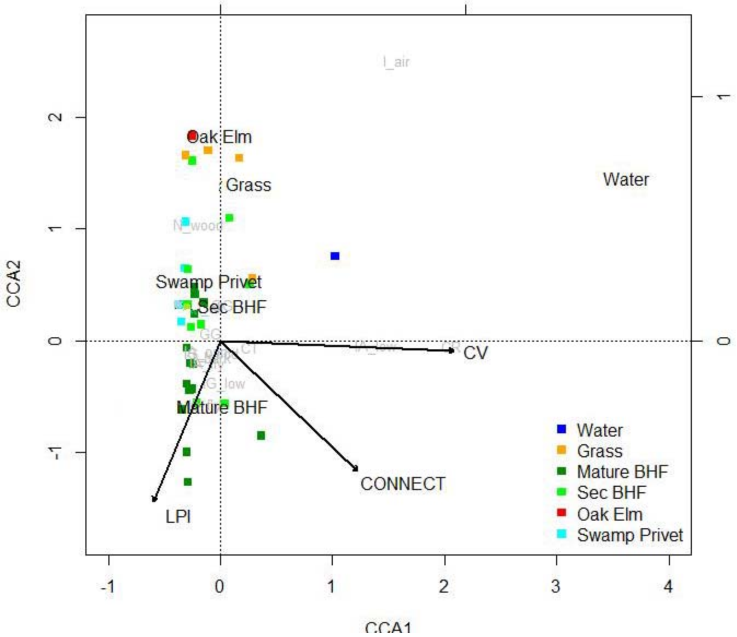


Figure 4.46. CCA plots showing relative position of sites, birds foraging guilds, and environmental variables for the 2010 bird community composition.

4.5.8 Identification of the Best Forest Bird Habitat in the GBC

Bird community composition associated with some of the secondary forests was similar to mature forests (Figure 4.33 for 2009 and Figure 4.34 for 2010). Therefore, late successional secondary forests increase functional connectivity among forest patches in the GBC. If habitat connectivity was enhanced by re-growth of secondary forests on fallow land, questions remain about when they start to function as effective corridors for target species or foraging guilds. To investigate when secondary forests start to have a habitat function for forest interior birds, minimum spanning tree (MST) analysis was performed. *Figure 4.47* displays a MST constructed for the 2010 bird census data, connected by the shorted dissimilarity in bird species composition among sites.

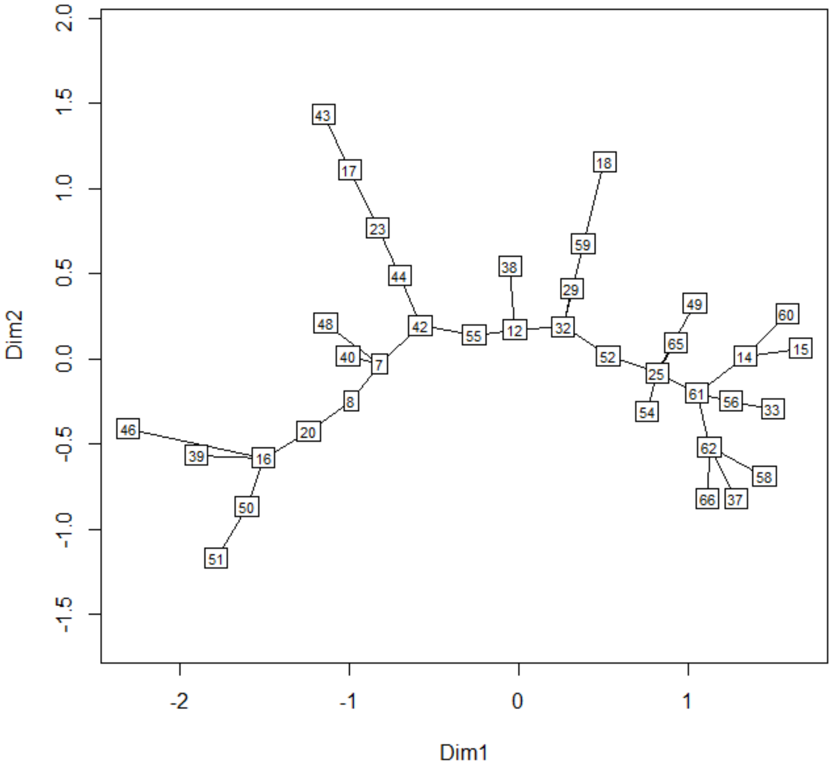


Figure 4.47. A MST plot showing minimum "dissimilarity" among sites constructed for the 2010 bird data. Sites were classified to two groups (mature forests on the right side vs. others on the left side) by a dash line. Physical characteristics of each site are summarized in the Appendix F.

Broadly speaking, this MST can be split into two clusters: sites with forest species vs. sites with more generalist species. It can be also inferred from this MST plot that bird species composition observed within the Oak-elm forest (site 29) is closely related to the species composition observed in mature bottomland hardwood forests. Moreover, species observed in late successional stands (sites 14, 60, and 49) had high similarities with mature bottomland hardwood forests. High similarities observed in late successional secondary forests and mature bottomland forests were consistent with CCA results. As we can see in Figure 4.33, sites 14, 60, and 49 were closely located to other mature bottomland hardwood forests.

Figure 4.48 shows sites with higher bird diversity measured by Shannon-Wiener index (H) for each sampling season of 2009 and 2010. Within the GBC, sites with higher diversity were found in larger extents of forests (forests containing sites 61 and 62, sites 12, 52, and 54, and the northwest corner of the GBC). The forest patch size distribution within GBC is displayed in Figure 4.49. Forest patches of interest are defined as contiguous pixels that contain trees taller than 10 m. CCA analysis did not identify forest patch size as a significant environmental descriptor to explain differences in bird community composition across habitat types (Section 4.5.5). Forest patch size, however, is positively correlated with forest extent and connectivity, both of which are critical factors in determining forest bird quality. In Figure 4.49 each color denotes that the pixel belongs to a patch of a given size class. Of all forest patches taller than 10 m in height (colored area in *Figure 4.49*), the ones in immediate riparian zones have the larger extent and seem contiguous indicating higher connectivity at landscape level. Thus, this map suggests that these large riparian patches have the highest habitat suitability for forest birds.

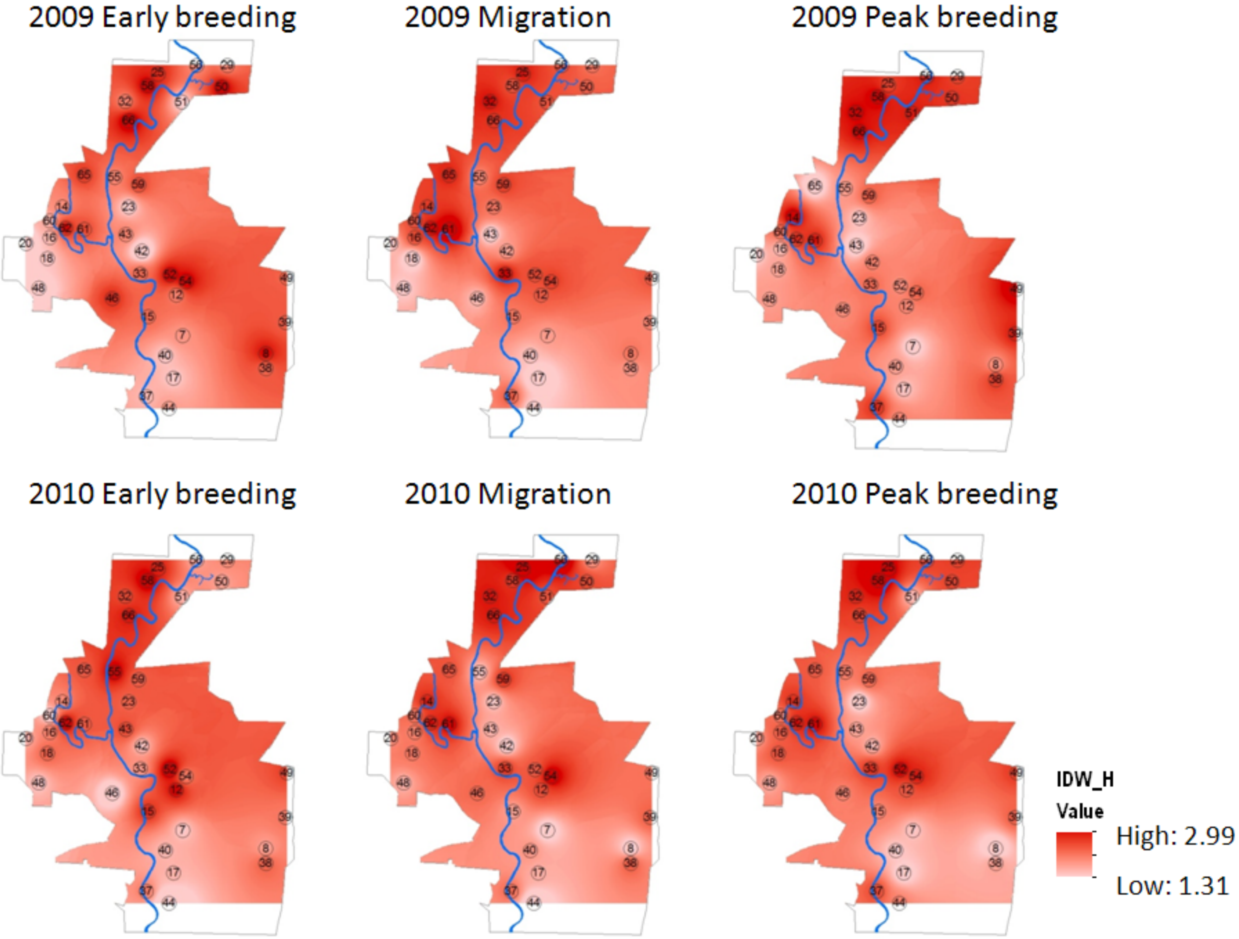


Figure 4.48. Maps showing sites with higher bird diversity throughout the sampling season of 2009 and 2010. Diversity was measured by Shannon-Wiener index (H). H values calculated for census points were spatially interpolated using Inverse Distance Weighting (IDW) method. Numbers on the map correspond with bird census points. Sites with higher diversity were indicated by darker shades.

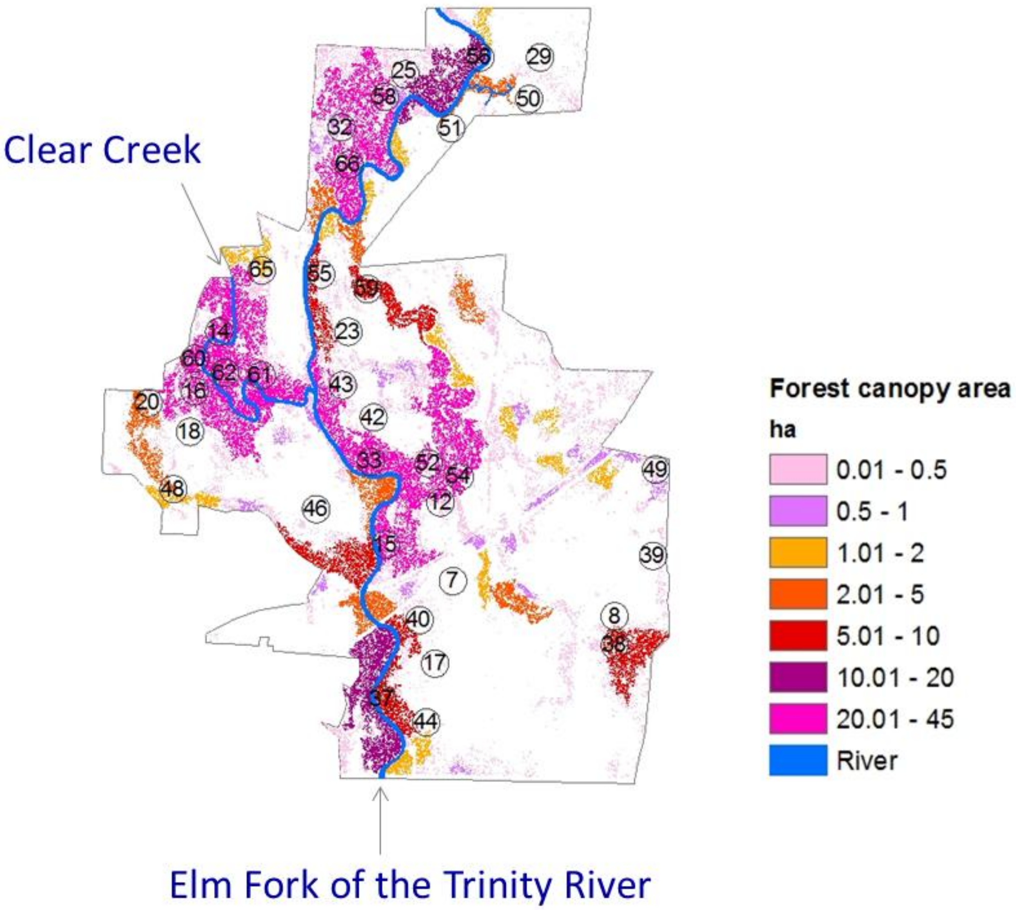


Figure 4.49. Forest patches size distribution. A color represents that the pixel belongs to forest patches (containing trees taller than 10 m in height) of given size. Physical characteristics of each site are summarized in the Appendix F.

CHAPTER 5

CONCLUSIONS AND SUGGESTIONS FOR FUTURE RESEARCH

This study demonstrated that three-dimensional habitat descriptions derived from integrated LiDAR and hyperspectral data serve as a powerful bird conservation tool that shows how the distribution of bird species relates to forest composition and structure at various scales. Accuracy assessments show that integrated LiDAR-hyperspectral increased the overall classification accuracy. Further improvement in accuracy will be achieved by finer spatial scale hyperspectral data, and more importantly, by using better noise reduction method of analysis of hyperspectral data. Specifically, spectral unmixing techniques combined with minimized noise fraction methods will greatly increase the overall accuracy in describing plant community composition.

Further, acquisition of full waveform LiDAR data allow a better delineation of understory structure, which was not obtained from the LiDAR data used for this analysis with a discrete return system. Accurate description of forest vertical structure will significantly increase our understanding of bird habitat niche requirements. Additionally, acquisition of leaf-on LiDAR data will be better suited for integration with hyperspectral data and assessing bird habitat during breeding season.

Spatial distribution patterns of local breeding birds within stands were highly influenced by vertical heterogeneity of vegetation surface. Some forest interior bird species show sills of bounded semivariance along riparian forests, indicating related abundances within the range. However, large-bodied raptors and generalist species were randomly distributed across the study site, indicating an independency from spatial arrangements of forests patches. Bird abundance across habitat types was well captured on their position in NMDS plots. Bird species composition observed in mature bottomland forests was closer to the one in some of the secondary forests, suggesting that late successional forests have similar habitat quality with mature bottomland forests.

For local breeding birds observed in the study site, four environmental variables: vertical heterogeneity of canopy surface within stands, connectivity to adjacent forest patches, largest forest patch index, and habitat (vegetation) types proved to be the most influential factors to determine bird community assemblages. Moreover, variation partitioning seem to suggest that the majority of explained variation in bird community composition is attributed to selected environmental variables themselves, and space alone seems to be a minor descriptor of bird assemblages at the scale of this study. Selected environmental variables still left approximately 50% of unexplained variation. Relatively high unexplained variation could be attributed to 1) inappropriate spatial scale to measure environmental gradients that occur at broader scale, 2) improper spatial scale to detect the abundance pattern of certain species, 3) deforestation history for agriculture and pastures might have eluded the effects of spatially structured environmental variables, 4) the disproportionate use of a habitat relative to its availability, and 5) environmental variables that were included might not be appropriate for this study. For future research, it is highly recommended to include more accurate forest vertical structure derived from a full waveform LiDAR. Segregation of vegetation layers using continuous LiDAR return will allow the measurements of shrub layer abundance or the size of sub-canopy gap, both of which are correlated with habitat requirements of many bird species.

Increasing sample size, particularly in upland oak-elm forests that generally occur at the late phase of bottomland hardwood succession, will improve our knowledge of how local breeding bird community composition responds to forest successional stages. A long-term bird population survey will also provide a better understanding of local and regional bird population dynamics. Further, investigating regional population sizes of each species could increase our knowledge of the mechanisms of the mismatch between habitat occupancy and suitable habitat saturation rates.

While there are similarities among habitat requirements of many bird species, habitat management to meet the specific needs of one species may or may not benefit other species. Although it is beyond the scope of this study to identify detailed habitat requirements for each individual bird species inhabiting various regions, generalizations can be made for preferred habitats for developing habitat management. This study has shown that an accurate multi-dimensional description of avian habitat serves as a powerful conservation tool that shows how bird community composition is related to forest stand structure at various spatial and temporal scales. Stratification of the species diversity data according to deforestation history, for example, recently clear-cut stands vs. relatively mature stands provides a method for future conservation planning.

Results also highlight the critical role of secondary forests to increase functional connectivity of forest patches. Previous studies have shown that human land use can place an irreversible impact on forest soils, plant community structure, and biodiversity (Dupouey et al., 2002). Further forest fragmentation seems inevitable around the study site due to close proximity to the Dallas-Fort Worth metropolitan complex. However, at least within the GBC, successive forest re-growth is occurring on fallow land. High similarities in bird species composition between late successional forests and mature forests (*Figure 4.33*, *Figure 4.34*, & *Figure 4.47*) signify the importance of secondary forests to enhance the quality of forest bird habitat. High similarities in bird species composition indicate that secondary forests in late successional stage may buffer edge effects around fragmented mature bottomland hardwood forests and facilitate movements of sensitive species across dispersed habitat patches and may contribute to increase the local populations' capacity to persist in such environment.

Within the GBC, the most suitable habitat for forest birds was found in immediate riparian zone along the Elm Fork of the Trinity River and Clear Creek (*Figure 4.48* and *Figure 4.49*). Forest area changes identified from historical maps (1879), aerial photos (1960, 1984, and 2000), and site specific land-use history (impoundments of reservoirs) suggest that some of the secondary forests currently at late successional stage including sites 12,16, 39, 55, and 60 on *Figure 4.49* were deforested sometime between 1960 and 1980. Identified areas from historical maps, located mostly on the water front, overlap with late successional forests which showed similar habitat functions with mature bottomland hardwood forests. Thus it can be inferred that regenerating forests surrounded by patches of mature bottomland hardwood forests within the GBC serve as suitable habitat for forest birds within 50-60 years.

It is, however, important to note that the current condition of forests (plant species composition, tree growth rates, and spatial distribution of plants) may be attributed to the hydrologic regime and land-use legacy before the constructions of the reservoirs nearby. Lake Lewisville located downstream of the GBC was originally built in 1927 and expanded in the1940’s and the 50’s (Cole-Jett, 2011). Upstream Lake Ray Roberts was impounded in 1987 (TPWD, 2013). These hydrologic alterations may have an impact on the future succession of this bottomland hardwood ecosystem.

APPENDIX A

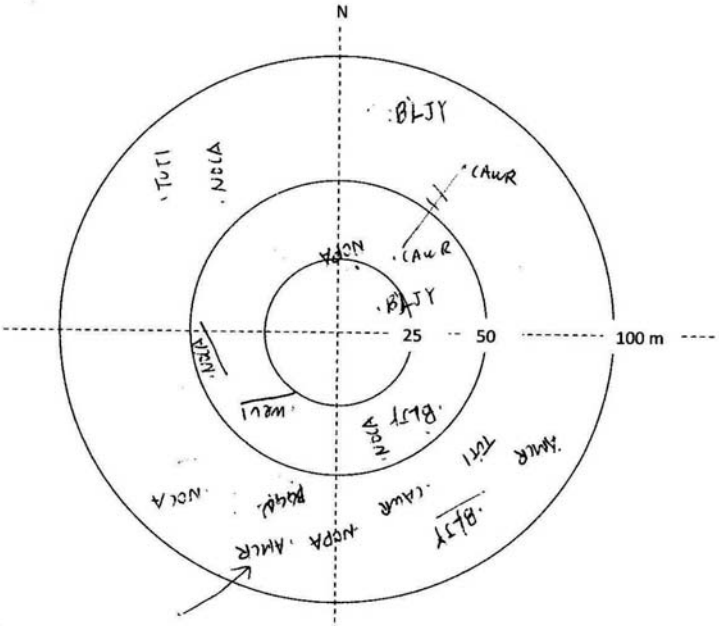
DATA SHEET USED FOR BIRD CENSUS

Point Count Location Mapping

②

Station # 63 Observer S.Y Date 4/11 Time 9:33-

Wind 5~7 mph Temperature 64 F Precipitation 0 Note ______



Mapping Symbols	
CODE	
○	Position of singing male
◌	Approximate position of singing male
○┤├○	Simultaneous song of 2 males
○→○	Known change in position
○⇢○	Assumed change in position
☆	Nest

APPENDIX B

GPS COORDINATES OF BIRD CENSUS POINTS. SPATIAL REFERENCE IS NAD 1983, UTM ZONE 14N. X IS LONGTITUDE AND Y IS LATITUDE

Site	x	y	Site	x	y
7	682758.47	3680858.45	43	681949.30	3682290.10
8	683955.76	3680627.02	46	681804.70	3681383.98
12	682666.92	3681432.40	44	682563.20	3679825.50
14	681042.84	3682681.40	48	680707.90	3681530.60
15	682268.47	3681125.79	49	684246.26	3681673.63
16	680865.91	3682245.71	50	683317.88	3684379.60
17	682628.05	3680254.53	51	682747.06	3684166.39
18	680831.20	3681949.48	52	682579.60	3681718.00
20	680526.32	3682162.50	54	682808.51	3681619.99
23	681991.32	3682679.82	55	681791.20	3683098.25
25	682410.70	3684561.10	56	682959.50	3684684.50
29	683398.12	3684683.64	58	682261.30	3684395.10
32	681940.53	3684173.44	59	682132.72	3682992.29
33	682156.21	3681730.59	60	680866.00	3682482.00
37	682239.89	3679996.00	61	681355.80	3682362.30
38	683952.16	3680365.06	62	681088.60	3682380.70
39	684226.40	3681043.20	65	681356.71	3683127.47
40	682513.58	3680572.51	66	681996.33	3683901.68
42	682177.00	3682054.00			

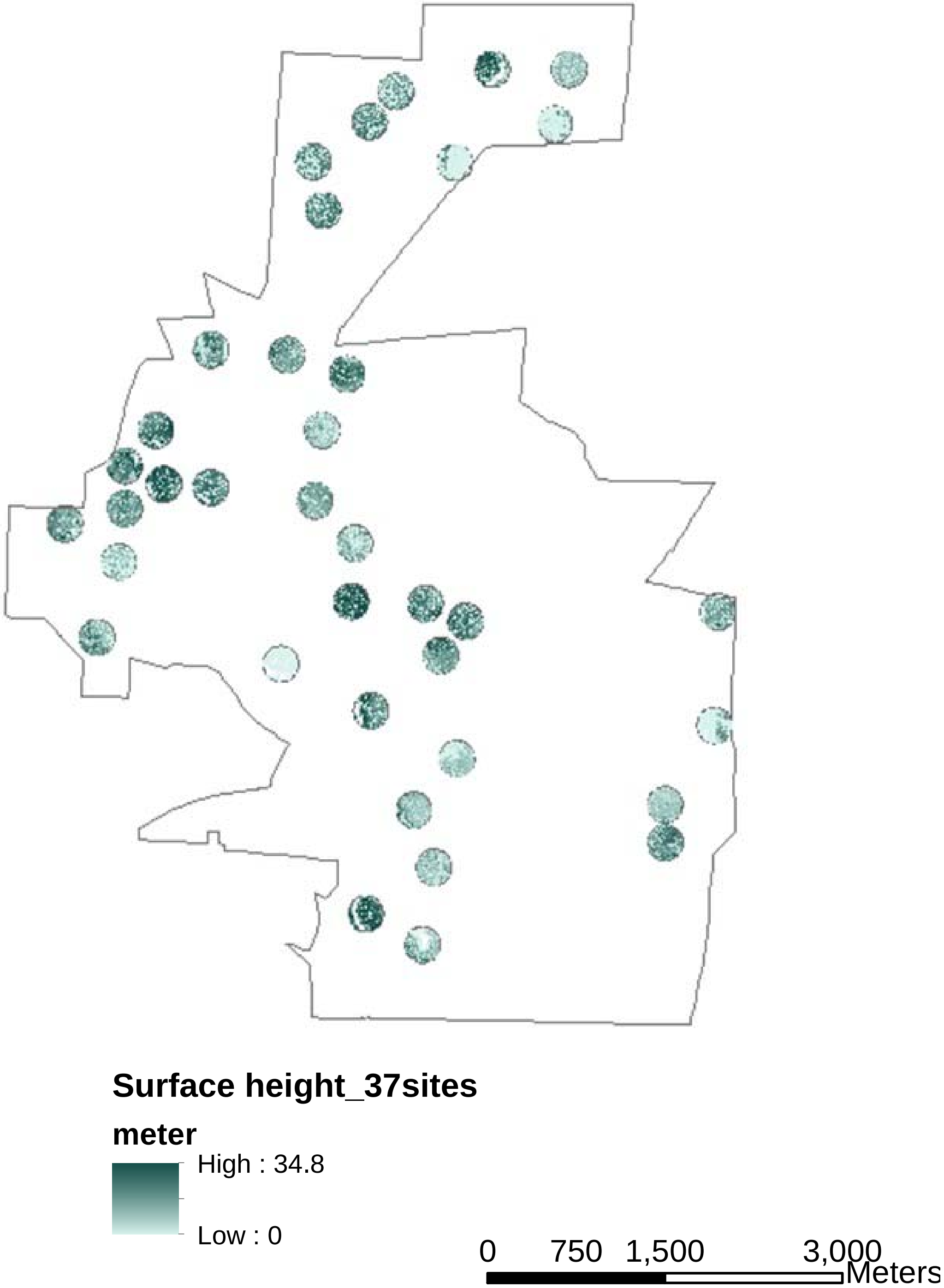
APPENDIX C

BIRD SPECIES LIST DETECTED DURING BREEDING SEASONS OF 2009 AND 2010

INBU	Indigo Bunting	Passerina cyanea	80	67
KILL	Killdeer	Charadrius vociferus	12	4
LESC	Lesser Scaup	Aythya affinis	1	0
LOWA	Louisiana Waterthrush	Seiurus motacilla	1	0
MODO	Mourning Dove	Zenaida macroura	16	4
NOCA	Northern Cardinal	Cardinalis cardinalis	375	455
NOFL	Northern Flicker	Colaptes auratus	5	19
NOHA	Northern Harrier	Circus cyaneus	1	0
NOMO	Northern Mockingbird	Mimus polyglottos	0	2
NOPA	Northern Parula	Parula americana	31	27
NOPI	Northern Pintail	Anas acuta	0	18
NOSH	Northern Shoveler	Anas clypeata	17	16
NRWS	Northern Rough-winged Swallow	Stelgidopteryx serripenn	0	5
OROR	Orchard Oriole	Icterus spurius	0	8
PABU	Painted Bunting	Passerina ciris	13	17
PIWO	Pileated Woodpecker	Dryocopus pileatus	1	21
PROW	Prothonotary Warbler	Protonotaria citrea	26	23
RBWO	Red-bellied Woodpecker	Melanerpes carolinus	111	105
RCKI	Ruby-crowned Kinglet	Regulus calendula	29	46
REVI	Red-eyed Vireo	Vireo olivaceus	70	57
RSHA	Red-shouldered Hawk	Buteo lineatus	54	40
RTHA	Red-tailed Hawk	Buteo jamaicensis	1	8
RTHU	Ruby-throated Hummingbird	Archilochus colubris	1	4
RWBL	Red-winged Blackbird	Agelaius phoeniceus	35	33
SAVS	Savannah Sparrow	Passerculus sandwichen	29	0
SNEG	Snowy Egret	Egretta thula	29	0
STFL	Scissor-tailed Flycatcher	Tyrannus forficatus	2	0
SUTA	Summer Tanager	Piranga rubra	14	20
SWTH	Swainson's Thrush	Catharus ustulatus	11	12
TNWA	Tennessee Warbler	Oreothlypis peregrina	0	1
TUTI	Tufted Titmouse	Baeolophus bicolor	115	185
TUVU	Turkey Vulture	Cathartes aura	53	36
WEKI	Western Kingbird	Tyrannus verticalis	1	0
WEVI	White-eyed Vireo	Vireo griseus	153	143
WIWA	Wilson's Warbler	Cardellina pusilla	0	1
WIWR	Winter Wren	Troglodytes troglodytes	1	0
WODU	Wood Duck	Aix sponsa	7	21
WOTH	Wood Thrush	Hylocichla mustelina	0	10
WTSP	White-throated *Sparrow*	Zonotrichia albicollis	3	9
YBCU	Yellow-billed Cuckoo	Coccyzus americanus	75	75
YCNH	Yellow-crowned Night-Heron	Nyctanassa violacea	1	0
YEWA	Yellow Warbler	Setophaga petechia	0	3
YRWA	Yellow-rumped Warbler	Dendroica coronata	19	37
YTWA	Yellow-throated Warbler	Setophaga dominica	0	2

APPENDIX D

HEIGHT OF VEGETATION IN 100 M RADIUS CIRCLES AROUND CENSUS POINTS


Surface height_37sites
meter
High : 34.8
Low : 0
0
750
1,500
3,000
Meters

APPENDIX E

CLOSE UP OF VEGETATION SURFACE HEIGHT OF A 100 M RADIUS CIRCLES AROUND EACH CENSUS POINT

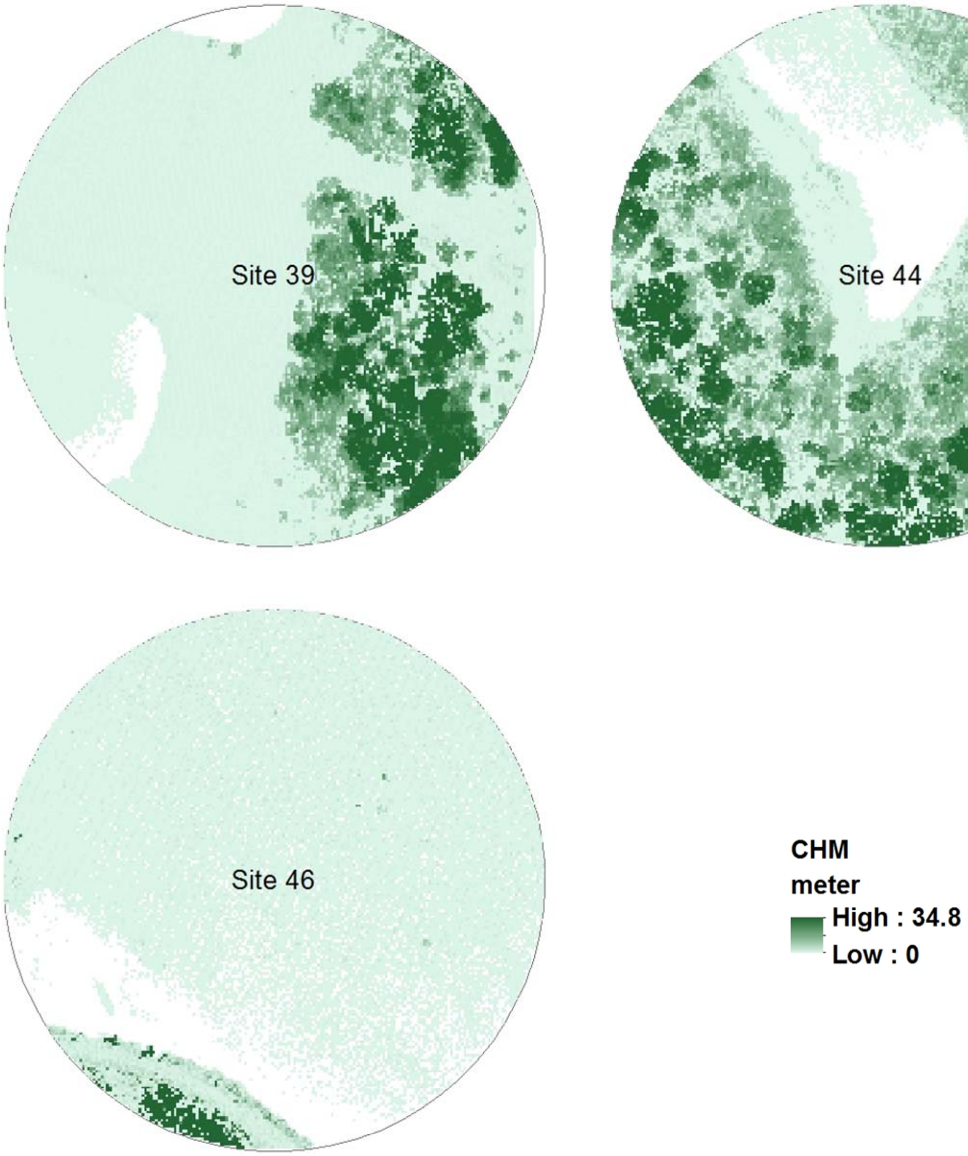


Figure E.1 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Water sites (sites 39, 44, & 46).

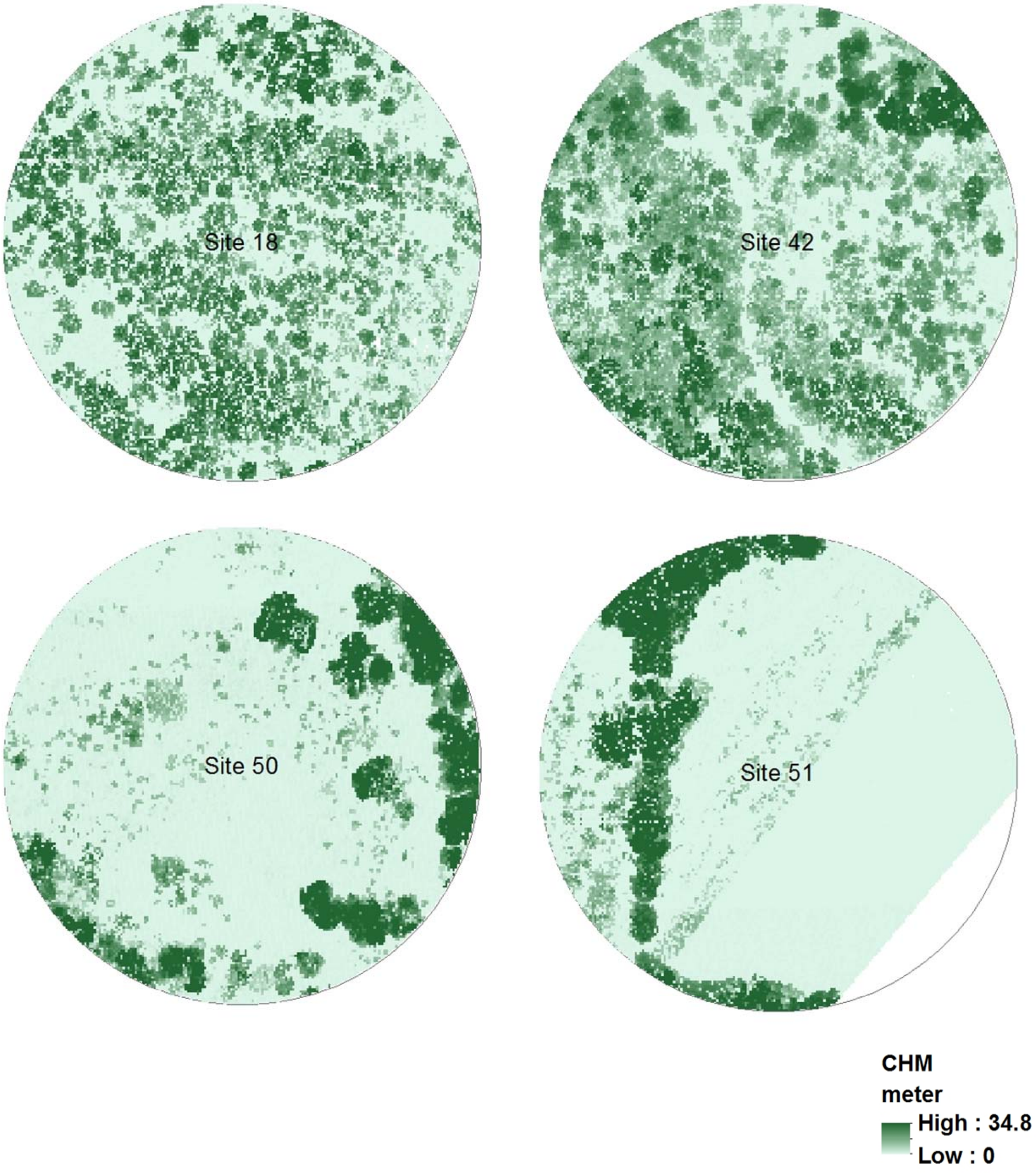


Figure E.2 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Grass_ herb sites (sites 18, 42, 50, & 51).

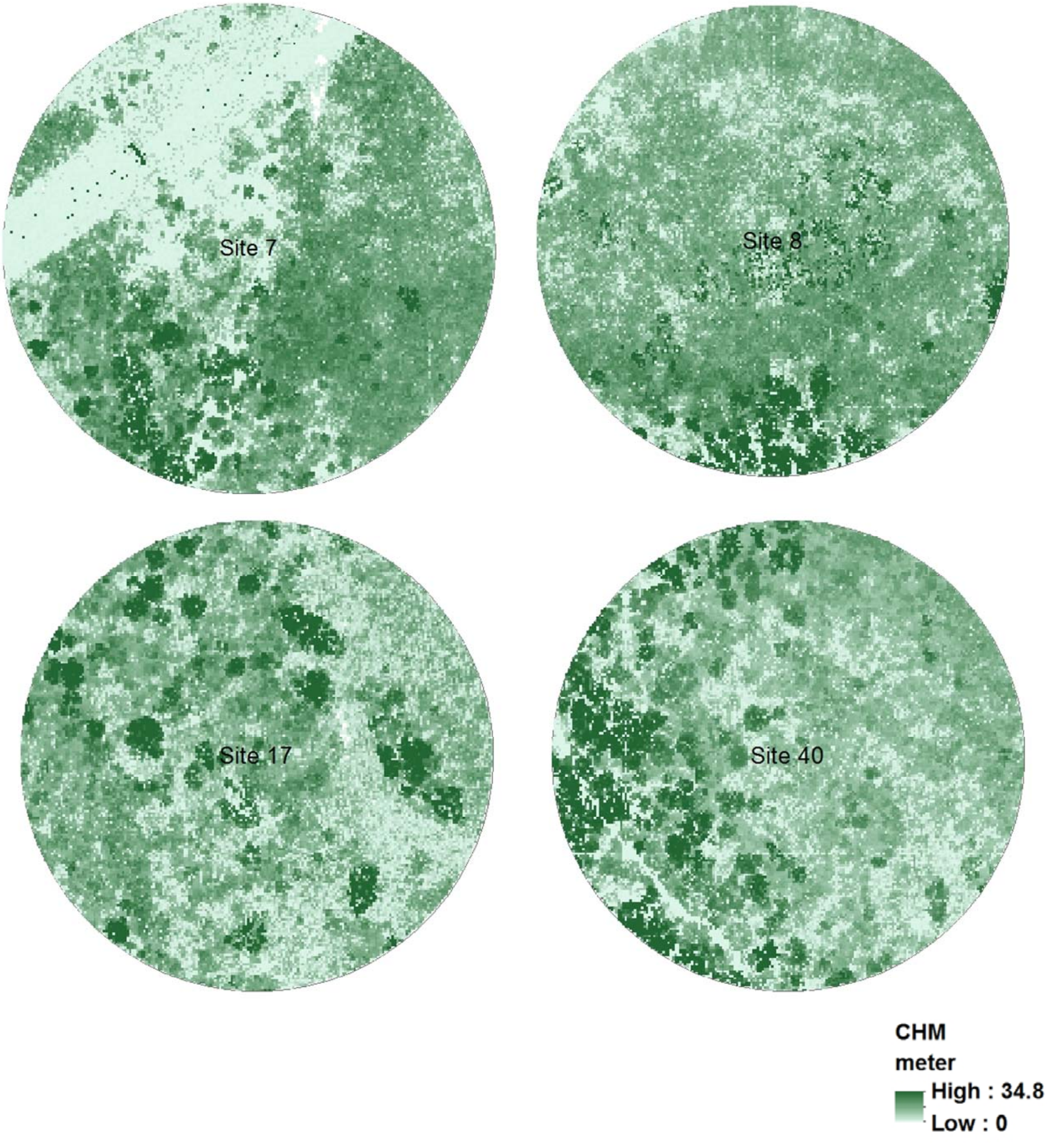


Figure E.3 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Wetland forest sites (sites 7,8,17, & 40).

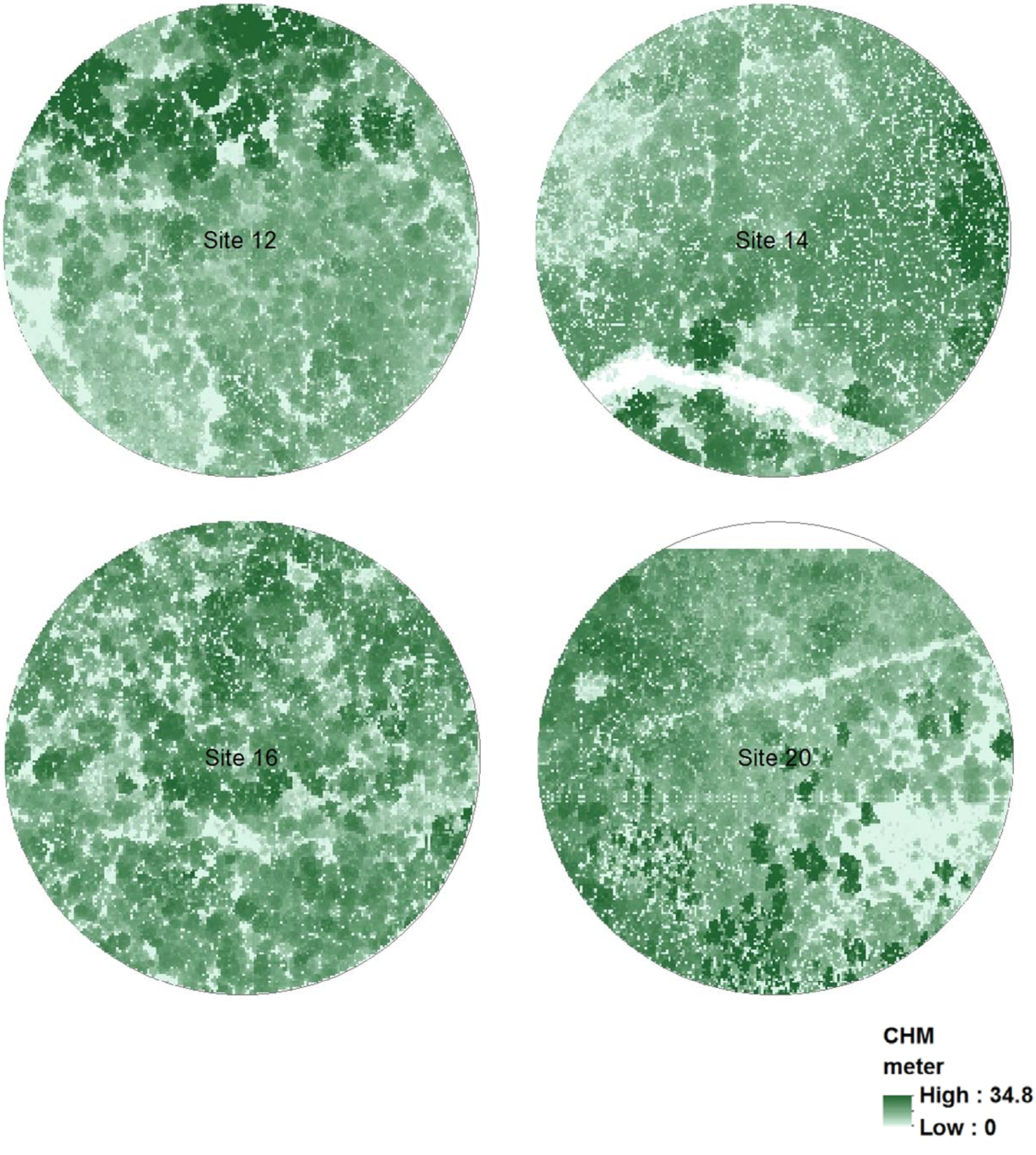


Figure E.4 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Sec BHF sites (sites 12, 14, 16, & 20).

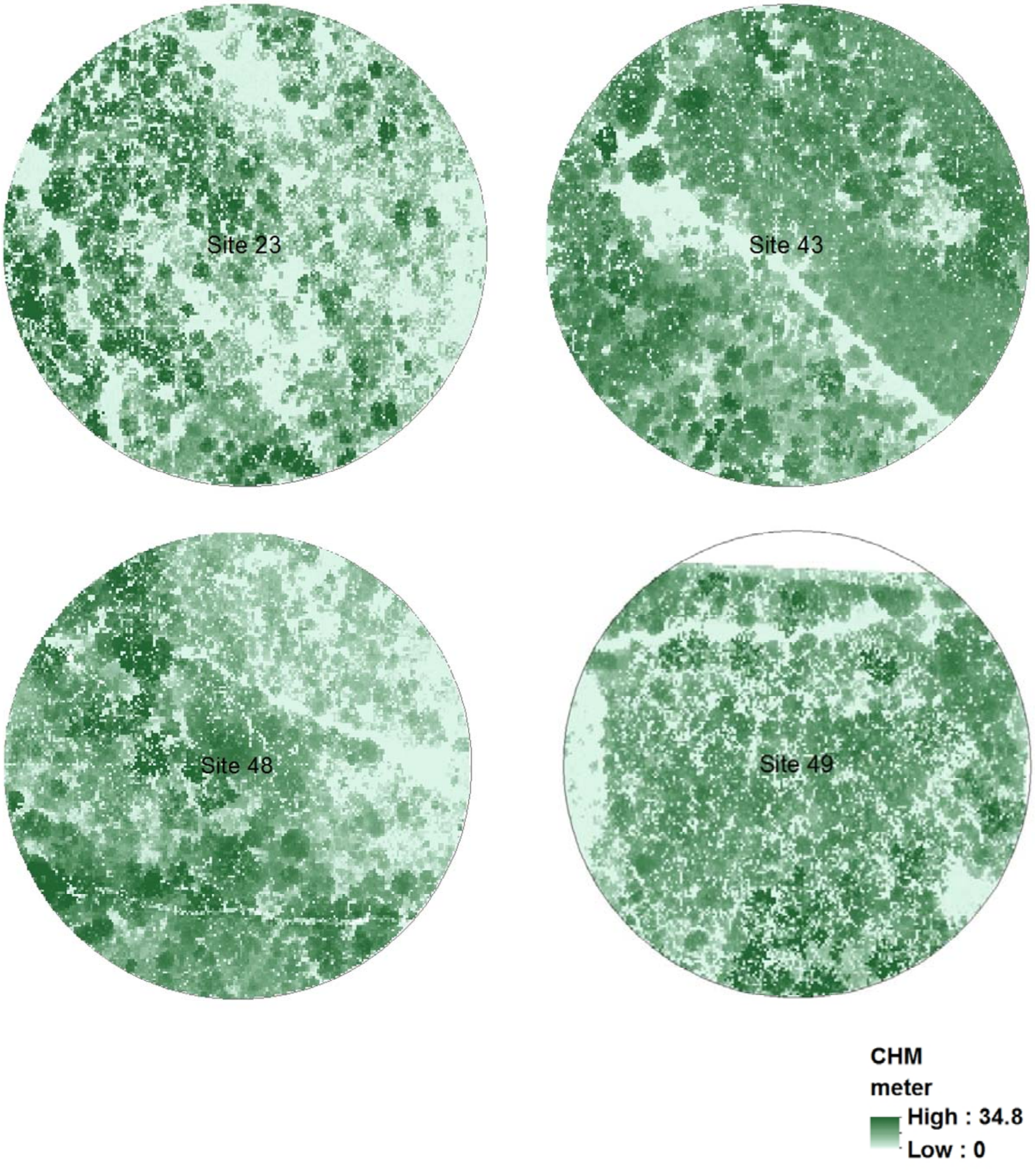


Figure E.5 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Sec BHF sites (sites 23, 43, 48, & 49).

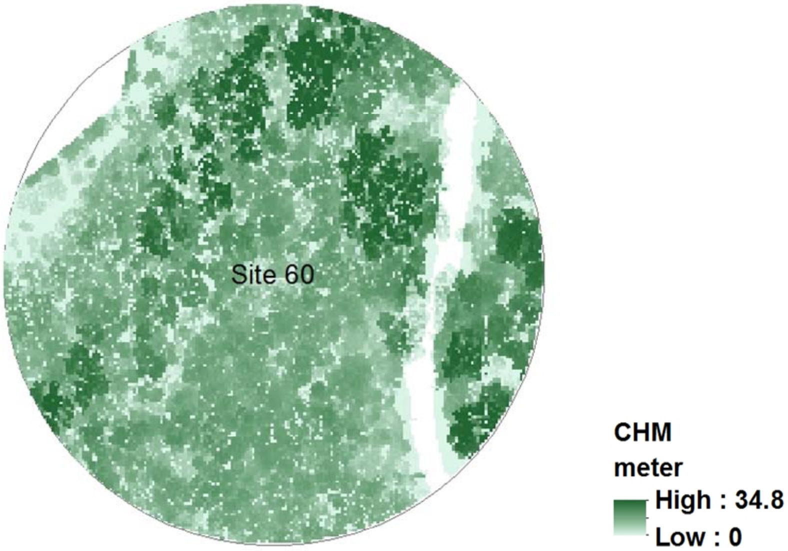


Figure E.6 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Sec BHF sites (site 60).

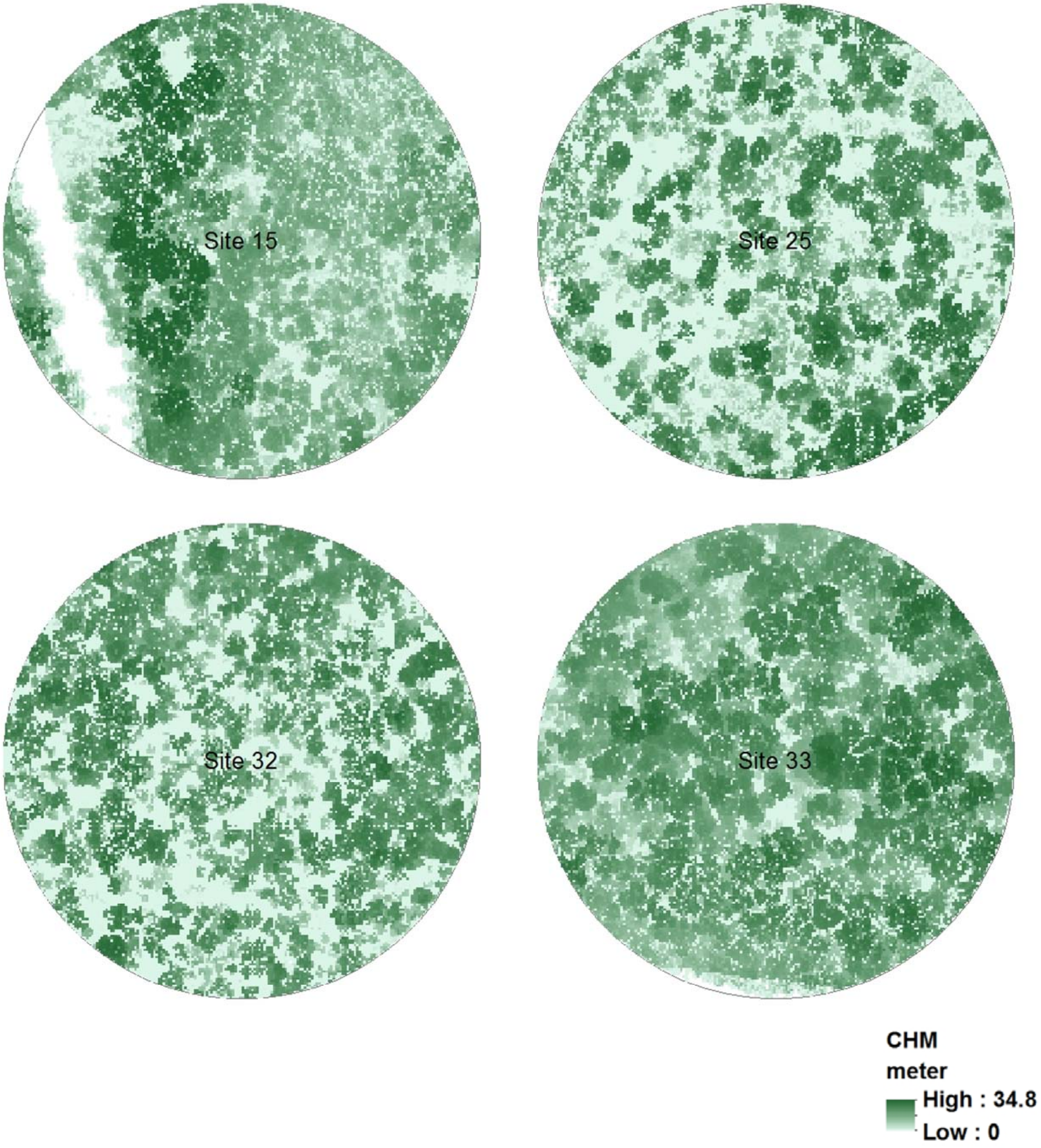


Figure E.7 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Mature BHF sites (sites 15, 25, 32, & 33).

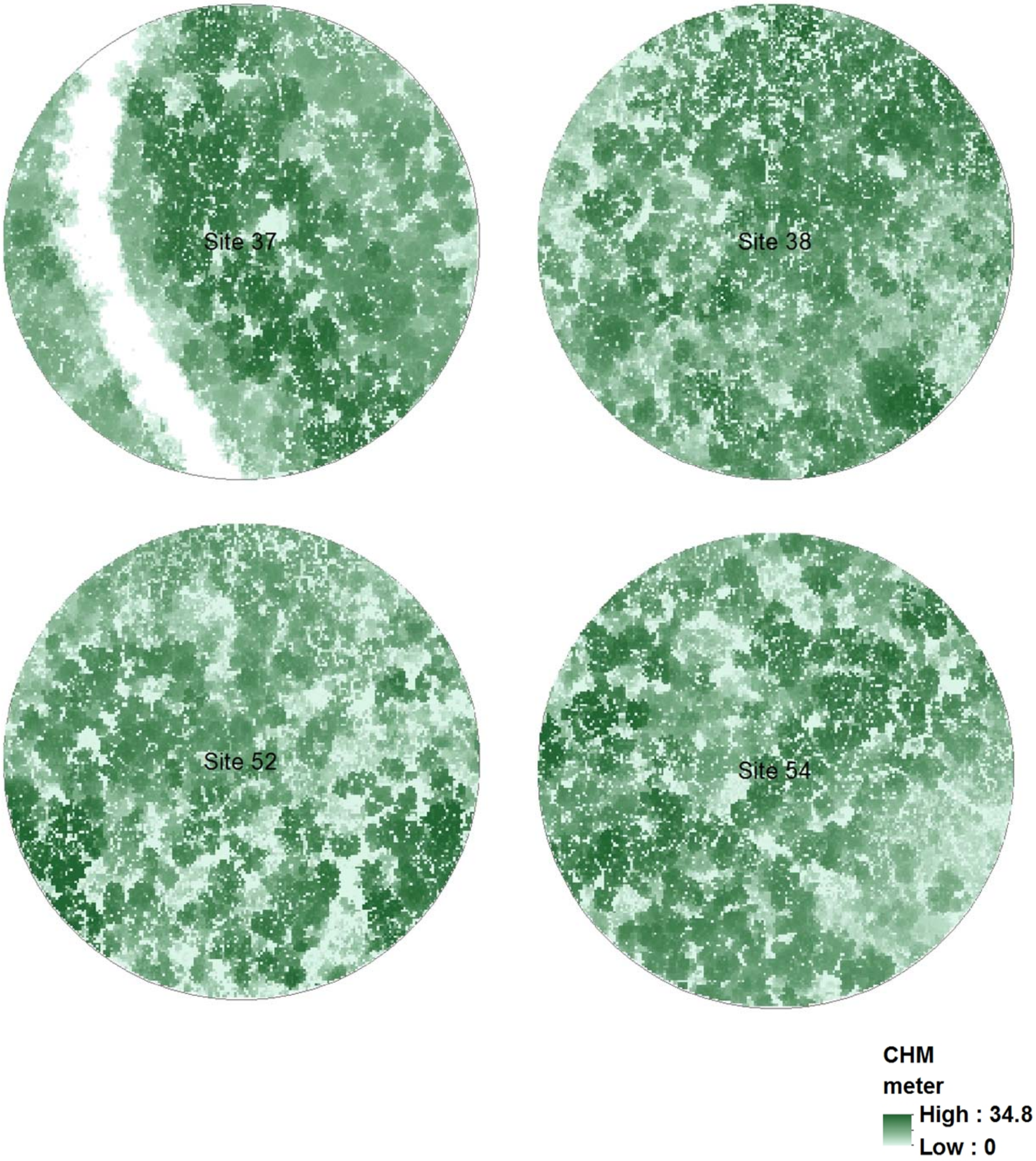


Figure E.8 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Mature BHF sites (sites 37, 38, 52, & 54).

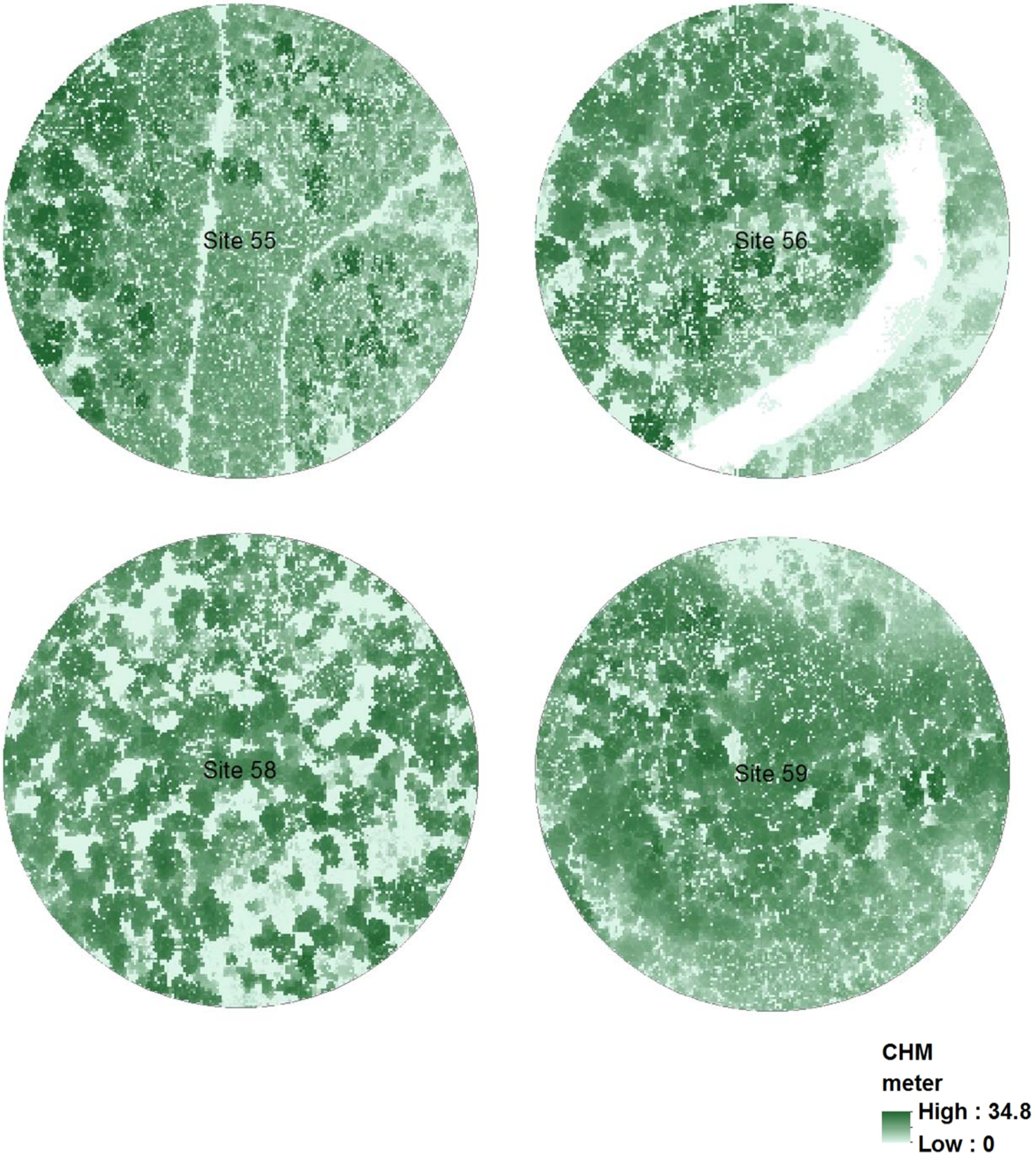


Figure E.9 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Mature BHF sites (sites 55, 56, 58, & 59).

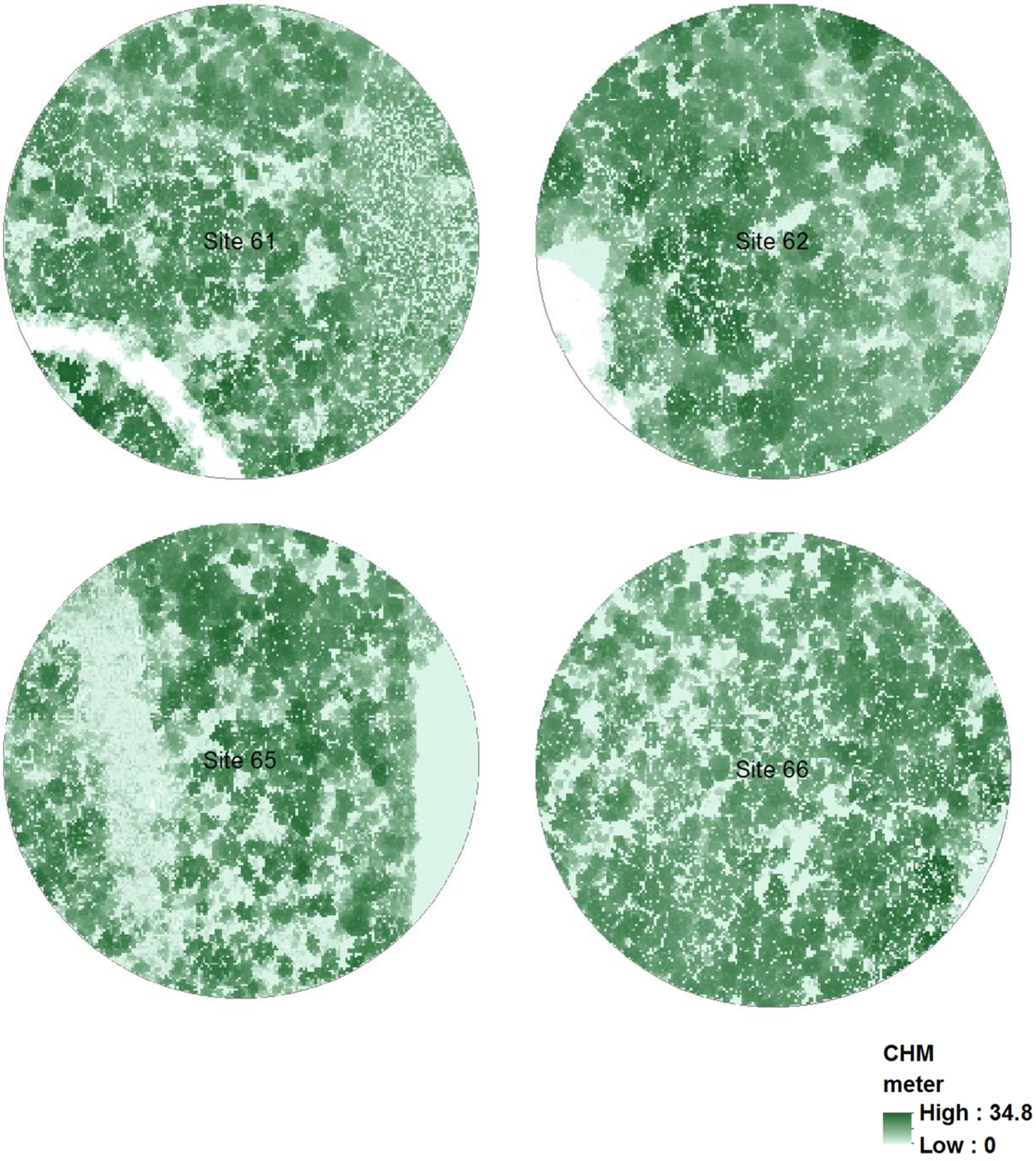


Figure E.10 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Mature BHF sites (sites 61, 62, 65, & 66).

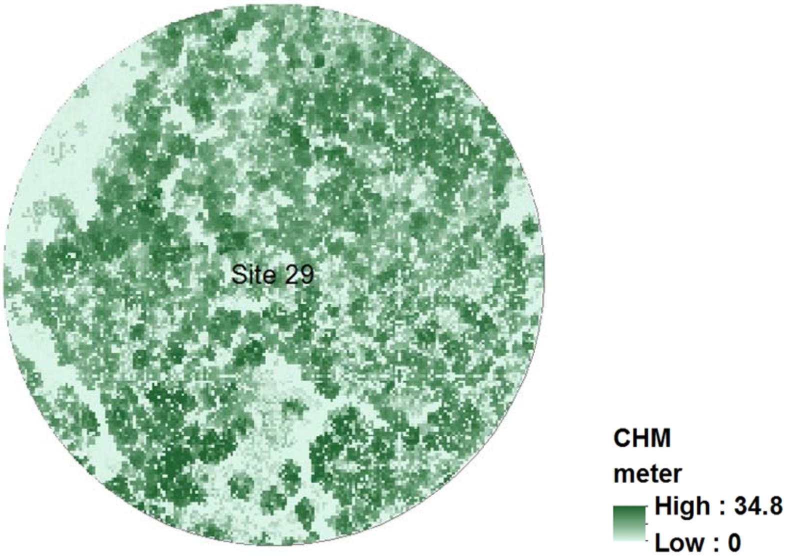
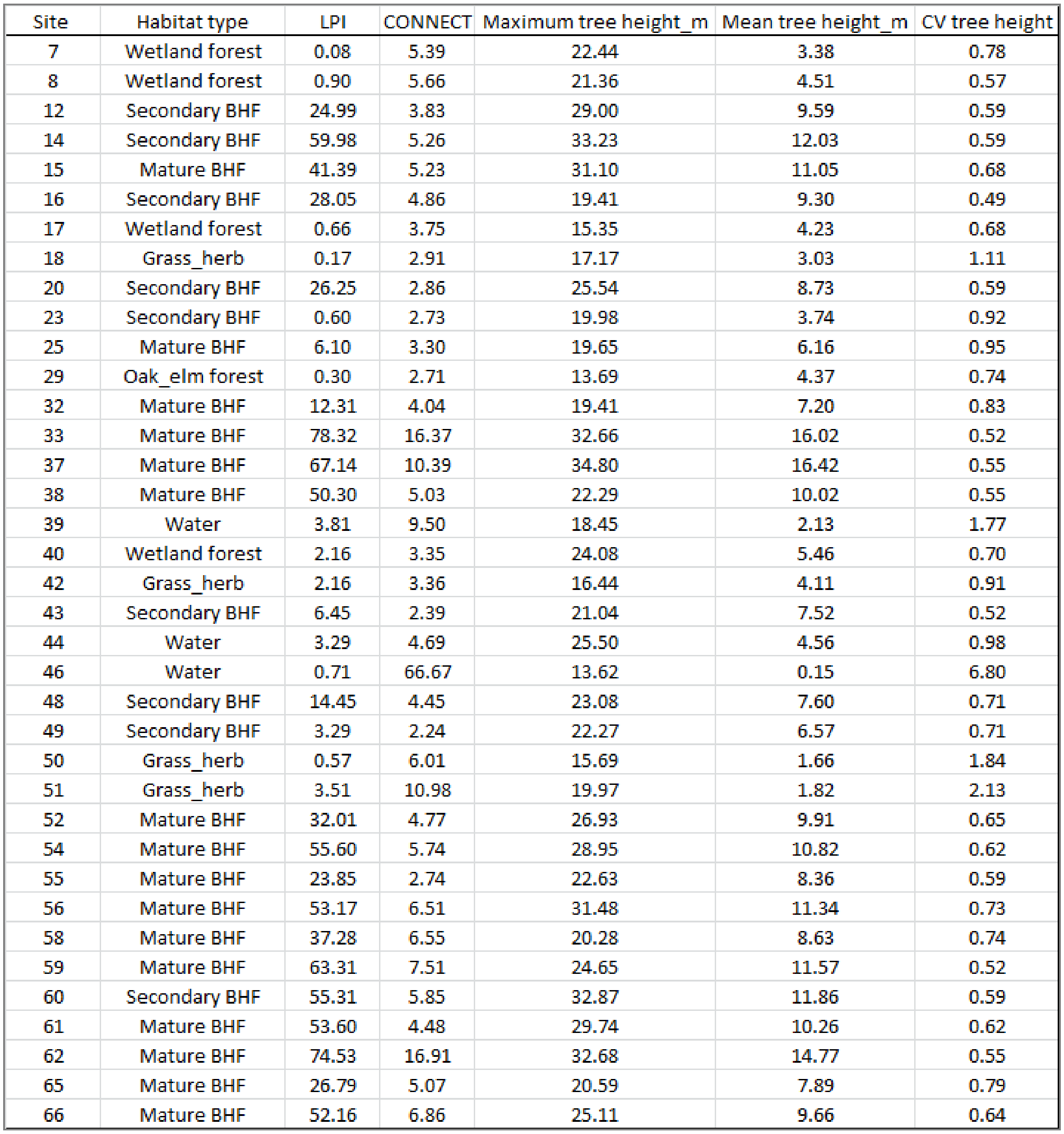


Figure E.11 Close up of vegetation surface height of a 100 m radius circles around each bird census point representing Oak-elm site (site 29).

APPENDIX F

ENVIRONMENTAL VARIABLES CALCULATED FOR 37 BIRD CENSUS POINTS AND ASSOCIATED CANOPY HEIGHT PROPERTIES



Site	Habitat type	LPI	CONNECT	Maximum tree height_m	Mean tree height_m	CV tree height
7	Wetland forest	0.08	5.39	22.44	3.38	0.78
8	Wetland forest	0.90	5.66	21.36	4.51	0.57
12	Secondary BHF	24.99	3.83	29.00	9.59	0.59
14	Secondary BHF	59.98	5.26	33.23	12.03	0.59
15	Mature BHF	41.39	5.23	31.10	11.05	0.68
16	Secondary BHF	28.05	4.86	19.41	9.30	0.49
17	Wetland forest	0.66	3.75	15.35	4.23	0.68
18	Grass_herb	0.17	2.91	17.17	3.03	1.11
20	Secondary BHF	26.25	2.86	25.54	8.73	0.59
23	Secondary BHF	0.60	2.73	19.98	3.74	0.92
25	Mature BHF	6.10	3.30	19.65	6.16	0.95
29	Oak_elm forest	0.30	2.71	13.69	4.37	0.74
32	Mature BHF	12.31	4.04	19.41	7.20	0.83
33	Mature BHF	78.32	16.37	32.66	16.02	0.52
37	Mature BHF	67.14	10.39	34.80	16.42	0.55
38	Mature BHF	50.30	5.03	22.29	10.02	0.55
39	Water	3.81	9.50	18.45	2.13	1.77
40	Wetland forest	2.16	3.35	24.08	5.46	0.70
42	Grass_herb	2.16	3.36	16.44	4.11	0.91
43	Secondary BHF	6.45	2.39	21.04	7.52	0.52
44	Water	3.29	4.69	25.50	4.56	0.98
46	Water	0.71	66.67	13.62	0.15	6.80
48	Secondary BHF	14.45	4.45	23.08	7.60	0.71
49	Secondary BHF	3.29	2.24	22.27	6.57	0.71
50	Grass_herb	0.57	6.01	15.69	1.66	1.84
51	Grass_herb	3.51	10.98	19.97	1.82	2.13
52	Mature BHF	32.01	4.77	26.93	9.91	0.65
54	Mature BHF	55.60	5.74	28.95	10.82	0.62
55	Mature BHF	23.85	2.74	22.63	8.36	0.59
56	Mature BHF	53.17	6.51	31.48	11.34	0.73
58	Mature BHF	37.28	6.55	20.28	8.63	0.74
59	Mature BHF	63.31	7.51	24.65	11.57	0.52
60	Secondary BHF	55.31	5.85	32.87	11.86	0.59
61	Mature BHF	53.60	4.48	29.74	10.26	0.62
62	Mature BHF	74.53	16.91	32.68	14.77	0.55
65	Mature BHF	26.79	5.07	20.59	7.89	0.79
66	Mature BHF	52.16	6.86	25.11	9.66	0.64

APPENDIX G

ESTIMATES OF VALUES OF DIRECTIONAL VARIOGRAMS FOR EACH BIRD SPECIES OBSERVED AT 37 CENSUS POINTS

Table G.5.1

Estimates of parameter values of directional variograms for each bird species observed at 37 census points in 2009.

2009													
		0^{o}			45^{o}			90^{o}			135^{o}		
Species	Sample size	nugget	range	sill	nugget	range	sill	nugget	range	sill	nugget	range	sill
NOCA	377	0	0	10	0	0	10	0	0	10	0	0	10
CACH	218	0	0	15	0	0	15	0	0	15	0	0	15
CAWR	195	0	500	20	0	0	20	0	0	20	0	0	20
WEVI	154	0	0	4	0	0	4	0	0	4	0	0	4
AMCR	152	0	4000	10	0	0	10	0	0	10	0	0	7
TUTI	117	0	0	10	0	0	10	0	0	10	0	4000	15
RBWO	37	0	0	4	0	0	2	0	0	4	0	0	4
BGGN	97	0	0	2	0	1000	2	0	0	2	0	0	2
INBU	77	0	2000	5	0	2000	5	0	0	1	0	0	NA
YBCU	72	0	0	0.3	0	0	0.3	0	0	0.1	0	0	NA
BHCO	71	0	1000	0.8	0	0	0.8	0	0	0.8	0	0	0.6
REVI	70	0	0	2	0	0	3	0	0	2	0	0	2
BLJA	55	0	0	1	0	0	1	0	0	1	0	0	1
RSHA	52	0	0	1	0	0	1	0	0	1	0	0	1
TUVU	51	0	0	1.8	0	0	1.8	0	0	1.8	0	0	1.8
DOWO	50	0	1000	1	0	0	1	0	0	1	0	0	1
GREG	46	0	0	2	0	0	2	0	0	NA	0	0	NA
EAWP	41	0	0	1	0	0	1.5	0	0	1.5	0	0	NA
GCFL	36	0	0	1	0	0	NA	0	0	1	0	0	1
NOPA	31	0	0	2	0	0	2	0	0	2	0	0	2
PROW	26	0	3000	1.5	0	0	1.5	0	0	1.5	0	0	1.5
RWBL	22	0	0	2	0	0	4	0	0	NA	0	0	1
GBHE	18	0	0	0.5	0	0	0.5	0	0	NA	0	0	0.5
CAEG	17	0	0	1	0	0	1	0	0	1	0	0	1
MODO	16	0	0	0.1	0	0	0.1	0	0	0.1	0	0	NA
BWTE	14	0	500	0.5	0	500	0.5	0	0	NA	0	0	2
SUTA	14	0	2000	1	0	0	0	0	0	1	0	0	1
AMRO	13	0	0	1	0	0	1	0	0	3	0	0	1
PABU	13	0	2000	1	0	2000	1	0	0	0.5	0	0	NA
BADO	12	0	0	0.5	0	0	0.5	0	0	0.5	0	0	0.5
KILL	12	0	0	0.2	0	0	0.2	0	0	NA	0	0	0.2
BASW	9	0	0	0.1	0	0	0.1	0	0	0.1	0	0	NA
GTGR	8	0	0	0.1	0	0	0.1	0	0	NA	0	0	0.1
EMPI	7	0	0	0.3	0	0	0.3	0	0	0.3	0	0	NA
WODU	7	0	0	1.5	0	0	1.5	0	0	1.5	0	0	NA
COHA	5	0	500	0.05	0	0	0.1	0	0	0.05	0	0	NA
EAPH	5	0	1000	NA	0	1000	0.1	0	1000	0.1	0	1000	0.1
NOFL	5	0	0	0.5	0	0	NA	0	0	0.5	0	0	0.5
BEKI	3	0	0	0.05	0	0	0.15	0	0	0.1	0	0	0.05
DICK	2	0	0	0.05	0	0	0.15	0	0	0.1	0	0	0.05
BAWW	1	0	0	0.4	0	0	0.4	0	0	NA	0	0	0.4
CCSP	1	0	0	0.2	0	0	0.4	0	0	NA	0	0	0.4
FISP	1	0	0	0.2	0	0	0.4	0	0	NA	0	0	0.4
LOWA	1	0	2000	1	0	0	NA	0	0	1	0	0	0.5
PIWO	1	0	2000	1	0	0	NA	0	0	1	0	0	0.5
RTHA	1	0	0	0.2	0	0	0.2	0	0	0.2	0	0	0.2
RTHU	1	0	0	0.1	0	0	0.1	0	0	0.1	0	0	0.1
SNEG	1	0	0	NA	0	0	10	0	0	10	0	0	10

Table G.2

Estimates of parameter values of directional variograms for each bird species observed at 37 census points in 2010.

2010													
		0°			45°			90°			135°		
Species	Sample size	nugget	range	sill	nugget	range	sill	nugget	range	sill	nugget	range	sill
NOCA	459	0	0	4	0	0	4	0	0	4	0	0	4
CAWR	307	0	0	7	0	0	7	0	0	7	0	0	7
CACH	273	0	0	3.5	0	0	3.5	0	0	3.5	0	0	3.5
TUTI	183	0	0	10	0	0	10	0	0	10	0	0	NA
AMCR	161	0	0	10	0	0	10	0	0	10	0	0	NA
WEVI	145	0	0	7	0	0	7	0	0	7	0	0	7
RBWO	113	0	1000	3	0	1000	3	0	0	3	0	0	2
YBCU	75	0	2500	11	0	0	5	0	0	0	0	0	NA
INBU	65	0	2500	8	0	2500	8	0	0	2	0	0	2
REVI	57	0	1000	3	0	0	3	0	0	3	0	0	3
EUST	55	0	2500	5	0	2500	5	0	0	2	0	0	NA
BGGN	54	0	0	7	0	0	NA	0	0	5	0	0	NA
DOWO	53	0	1500	3	0	0	1	0	0	1	0	0	NA
BLJA	50	0	0	2	0	0	2	0	0	2	0	0	2
RSHA	37	0	0	1	0	0	1	0	0	NA	0	0	1
GCFL	37	0	2500	5	0	2500	5	0	0	0.5	0	0	NA
TUVU	34	0	0	2	0	0	2	0	0	2	0	0	NA
NOPA	27	0	0	NA	0	0	2	0	0	2	0	0	2
EAWP	26	0	0	4	0	0	4	0	0	4	0	0	4
AMRO	25	0	0	7	0	0	7	0	0	10	0	0	5
BHCO	25	0	0	NA	0	0	20	0	0	20	0	0	20
PROW	23	0	0	0.5	0	0	1.5	0	0	0.5	0	0	1
PIWO	21	0	0	0.02	0	0	0.04	0	0	0.04	0	0	0.04
WODU	21	0	0	0.2	0	0	0.2	0	0	NA	0	0	0.2
SUTA	20	0	3000	2	0	3000	4	0	0	0	0	0	1
NOFL	19	0	0	0.1	0	0	0.1	0	0	0.1	0	0	NA
PABU	17	0	0	1	0	0	1	0	0	0.25	0	0	NA
GREG	16	0	0	7	0	0	7	0	0	NA	0	0	7
GBHE	15	0	0	0.5	0	0	NA	0	0	1	0	0	1
RWBL	15	0	0	0	0	0	3	0	0	NA	0	0	3
BADO	13	0	0	0.3	0	0	0.3	0	0	0.8	0	0	0.3
AMCO	11	0	0	0.5	0	0	0.5	0	0	NA	0	0	0.5
WOTH	10	0	0	5	0	0	NA	0	0	5	0	0	5
BWTE	8	0	0	NA	0	0	0	0	0	NA	0	0	1
EMPI	8	0	0	0.2	0	0	0.2	0	0	0.2	0	0	NA
OROR	8	0	2000	0.4	0	0	0.4	0	0	0.1	0	0	NA
RTHA	7	0	0	0.2	0	0	NA	0	0	0	0	0	0
FISP	7	0	0	0.4	0	0	0.4	0	0	NA	0	0	0.4
BEVI	6	0	1000	0.4	0	0	0.5	0	0	0	0	0	NA
EABL	5	0	2000	1	0	2000	1	0	2000	1	0	2000	1
NRWS	5	0	0	0.5	0	0	0.5	0	0	NA	0	0	0.5
BLVU	4	0	0	0.1	0	0	0.1	0	0	0.1	0	0	NA
KILL	4	0	0	0.1	0	0	0.1	0	0	0.1	0	0	NA
MODO	4	0	0	0.1	0	0	0.2	0	0	0.1	0	0	NA
YEWA	4	0	0	0.1	0	0	0.1	0	0	0.1	0	0	0.1
RTHU	4	0	0	0.2	0	0	0.2	0	0	0.2	0	0	0.2
CHSP	4	0	1000	0.4	0	1000	0.4	0	1000	0.4	0	1000	0.4
BEKI	3	0	0	0	0	0	0.5	0	0	0.5	0	0	0.5
BASW	3	0	0	0.2	0	0	0.2	0	0	0.2	0	0	NA
EAPH	3	0	0	NA	0	0	0.1	0	0	NA	0	0	0.1
GTGR	2	0	0	0.05	0	0	0.05	0	0	NA	0	0	0.05
NOMO	2	0	0	0.05	0	0	NA	0	0	0	0	0	0.05
COHA	2	0	0	0.05	0	0	0.1	0	0	0.05	0	0	NA
YTWA	2	0	1000	0.05	0	0	0.05	0	0	0.05	0	0	NA
EAKI	1	0	0	0.05	0	0	0.05	0	0	0.05	0	0	NA
WEKI	1	0	0	0	0	0	0.05	0	0	0	0	0	0.05

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