Influence Of Landscape Spatial Patterns And Land Use Planning On Grassland Bird Habitat Occupancy In Chester County, Pennsylvania

M. Zoe Warner
University of Pennsylvania, zoemiros@hotmail.com

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Influence Of Landscape Spatial Patterns And Land Use Planning On Grassland Bird Habitat Occupancy In Chester County, Pennsylvania

Abstract
Throughout their range, North American grassland birds as a group have shown steeper and more widespread population declines than any other avian guild. Grassland bird decline corresponds to habitat loss, landscape fragmentation, agricultural intensification, and changes in land management. In areas where farmland is being converted, the presence or absence of grassland birds can be used to assess the availability of habitat across the landscape. Chester County in southeastern Pennsylvania has historically had an agricultural economic base, but since the 1980s, steady population growth and economic diversification have reduced the amount of available agricultural land in the county. In the last decade, efforts have been underway to slow land conversion and conserve working landscapes. Because Chester County has maintained its working landscapes despite development pressures, the county provides an opportunity to examine the effects of land use change and agricultural preservation on grassland bird occurrence. This dissertation uses fixed-radius point counts of six focal bird species on agricultural land in Chester County to evaluate the influence of landscape spatial patterns and land use planning outcomes on grassland bird habitat occupancy. Separate habitat models are generated for the focal guild and individual species to predict habitat occupancy within a 750 square kilometer area in Chester County's agricultural belt. A landscape diagnosis of spatial patterns quantified by landscape metrics computed for discrete landcover types is developed to assess current landscape spatial patterns and proposed land use initiatives in the county as they relate to grassland bird conservation. The model outcomes and the landscape analysis indicate the grass-cropland network within the study area could provide suitable habitat for grassland birds. The study area has a relatively high level of spatial integrity for a county that has undergone rapid development and a core mixed agricultural area remains. These factors have important implications for the persistence of agricultural land uses and habitat availability for grassland birds.

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INFLUENCE OF LANDSCAPE SPATIAL PATTERNS AND LAND USE PLANNING ON GRASSLAND BIRD HABITAT OCCUPANCY IN CHESTER COUNTY, PENNSYLVANIA

M. Zoë Warner

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in

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Supervisor of Dissertation

__________________________________________
Thomas L. Daniels
Professor of City and Regional Planning

Graduate Group Chairperson

__________________________________________
Eugenie L. Birch
Lawrence C. Nussdorf Professor of Urban Research & Education

Dissertation Committee

Thomas L. Daniels, Professor of City and Regional Planning
C. Dana Tomlin, Professor of Landscape Architecture
Richard J. Horwitz, Professor, Department of Biodiversity, Earth & Environmental Science; Senior Scientist, Fisheries Section Leader, Ruth Patrick Chair of Environmental Sciences, Academy of Natural Sciences of Drexel University
To Sophie-Bird and Will-Bear,
seeing the world through your young, hopeful eyes
inspires me to seek the possibilities
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ABSTRACT

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M. Zoë Warner

Thomas L. Daniels

Throughout their range, North American grassland birds as a group have shown steeper and more widespread population declines than any other avian guild. Grassland bird decline corresponds to habitat loss, landscape fragmentation, agricultural intensification, and changes in land management. In areas where farmland is being converted, the presence or absence of grassland birds can be used to assess the availability of habitat across the landscape. Chester County in southeastern Pennsylvania has historically had an agricultural economic base, but since the 1980s, steady population growth and economic diversification have reduced the amount of available agricultural land in the county. In the last decade, efforts have been underway to slow land conversion and conserve working landscapes. Because Chester County has maintained its working landscapes despite development pressures, the county provides an opportunity to examine the effects of land use change and agricultural preservation on grassland bird occurrence. This dissertation uses fixed-radius point counts of six focal bird species on agricultural land in Chester County to evaluate the influence of landscape spatial patterns and land use planning outcomes on grassland bird habitat occupancy. Separate habitat models are generated for the focal guild and individual species to predict habitat occupancy within a 750 square kilometer area in Chester County’s agricultural belt. A landscape diagnosis of spatial patterns quantified by landscape metrics computed for discrete land cover types is
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CHAPTER 1 – GRASSLAND BIRDS AND THEIR RELATIONSHIP TO THE LANDSCAPE

All the fields along
I can hear the song
Of the meadow lark,
As she flits and flutters,
And laughs at the thunder when it mutters.

--Paul Laurence Dunbar, The Meadow Lark

History and Status of Grassland Birds

Grassland birds are broadly defined as “any species that has become adapted to and reliant on some variety of grassland habitat for part or all of its life cycle,” including breeding—both nesting and feeding—migration, or wintering (Vickery et al. 1999, 5). Although grassland birds once “prospered” in a landscape cultivated by human hands, Harold Mayfield (1989, 47) noted the “quiet decline” of these birds throughout their range. In the thirty years since Mayfield’s warning, the plight of grassland birds has not attracted as much attention as declines among forest dwelling bird species, especially in the eastern United States where grassland bird species are often misrepresented as non-native migrants.

Grassland birds are most commonly associated with the Great Plains (Knopf 1994) and the Midwest (Herkert 1995), which historically have been centers of diversity and abundance among this guild. Yet, grassland birds have also been a part of Eastern U.S. landscapes for centuries (Vickery and Dunwiddie 1997). The myth of the pre-Columbian and Colonial eastern landscape as a swath of uninterrupted forest stretching out to the plains of the Midwest contrasts with the reality of the “mosaic of forests and fields in varying degrees of succession” (Patterson and Sassaman 1988, 115) that
characterized the landscape. Many disturbance-dependent bird species have long been part of the Eastern avifauna and are not just recent migrants that colonized the eastern grasslands from more suitable western habitats. Pollen deposits and skeletal remains suggest there were natural prairies and savannas on the East Coast that would have been suitable for grassland species (Askins 2002). Prior to European settlement, Native Americans used fire to clear land for agriculture and maintained grasslands to facilitate hunting (Patterson and Sassaman 1988). When European settlers arrived, they opened more of the forests as they cleared land for crops and grazing and introduced grass species such as bluegrass (*Poa pratensis*) and white clover (*Trifolium repens*) (Cronon 1983). Their agricultural practices were compatible with grassland bird breeding and foraging requirements, and grassland bird populations increased in response (Askins 1999; Hunter et al. 2001). In addition to human changes to the land, beavers (*Castor canadensis*) would abandon dams leaving behind “beaver meadows,” which are grasslands or meadows with shrubby vegetation that are suitable nesting sites for Savannah Sparrows (*Passerculus sandwichensis*) and Bobolinks (*Dolichonyx oryzivorus*) (Askins 1999).

Through the combination of pre- and post-European forest clearing, there was habitat to support grassland bird guilds, and evidence shows they were common and abundant in the East. Several eastern populations of grassland bird species are distinct enough from western populations to suggest eastern and western populations have been reproductively isolated for thousands of years (Brennan and Kuvlesky 2005). In the East, observations of grassland birds may go back to the sixteenth century, but it is difficult to confirm because the descriptions are incomplete and old-world species names were
assigned to North American birds. By the 1800s, however, there was unambiguous documentation of bird species. Early ornithologists Alexander Wilson and John James Audubon described eastern bird species and documented the presence of Bobolinks, Grasshopper Sparrows (*Ammodramus savannarum*), Eastern Meadowlarks (*Sturnella magna*), Upland Sandpipers (*Bartramia longicauda*), and other common grassland species (cited in Askins 2002).

Though many species were well established by the nineteenth century, expanded “grassland” habitat enabled more prairie species to move eastward from the Midwest (Askins 2002); among these migrants was the prairies subspecies of the Horned Lark (*Eremophila alpestris praticola*), which expanded its range eastward through the late 1800s (Askins 2002; Wilson et al. 2012) and was described as breeding widely in the East in the early 1900s (Harlow 1918).1 In Pennsylvania, species associated with natural open areas were widespread by the late nineteenth century, and bird species associated with the more western prairies were using agricultural lands (Wilson et al. 2012).

With the expansion of farmlands and hayfields, there was little change in grassland populations through the 1950s (Warner 1994). But by the 1960s, population trends had changed. Throughout their entire range, grassland birds as a group have “shown steeper, more consistent, and more geographically widespread declines than any other behavioral or ecological guild” (Knopf 1994, 251). Data from the North American Breeding Bird Survey, which began in 1966, show grassland obligate species declined 37.8 percent between 1968 and 2011, with negative population estimate trends for 75

---

1 The northern subspecies (*E. a. alpestris*) was known to winter in Pennsylvania but breeding populations became established later (Wilson et al. 2012)
percent of grassland birds (Sauer et al. 2013). Since 1966, 15 of the 19 grassland and savanna species that breed in eastern North America have been declining, with some species, such as Grasshopper Sparrows and Eastern Meadowlarks, experiencing rapid population changes (Askins 2002).

Several factors have contributed to the decline in grassland species. These factors include: the destruction and degradation of native grassland habitats (Knopf 1994; Vickery and Dunwiddie 1997; Peterjohn and Sauer 1999); habitat fragmentation; afforestation that reduces the number of early successional fields and old-field habitats in the landscape; nest parasitism (Peterjohn and Sauer 1999; Brennan and Kuvlesky 2005), and fire suppression (Vickery et al. 2005). In the Northeast, grasslands are commonly lost when abandoned farmland reverts to forest or is developed for residential and commercial uses (Askins 1993; Goodrich et al. 2002). This contributes to increased landscape fragmentation, which can affect grassland bird habitat occupancy and community structure, especially among area sensitive species (Herkert 1994).

Species decline also corresponds to agricultural intensification and changes in management practices to increase efficiency. These changes include: converting grasslands, shrublands, and wetlands into crop production (North American Bird Conservation Initiative, U.S. Committee 2013); moving to an intensive row cropping model that reduces the landscape structure and involves the use of synthetic fertilizer and pesticides (Warner 1994); converting hayfields to alfalfa (Medicago sativa) that produce

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2 Between 2008 and 2011, over 9.3 million hectares (23 million acres) of non-agricultural grasslands, shrublands, and wetlands were converted to crop production (North American Bird Conservation Initiative, U.S. Committee 2013).
higher and more frequent yields (Askins et al. 2007); and mowing hayfields earlier and more frequently, which increases total seasonal production, but also coincides with breeding activity, jeopardizing ground nests and fledglings (Askins 1993).

With their specific habitat requirements for breeding and foraging, bird species demonstrate sensitivity to landscape properties (McGarigal and McComb, 1995). Grassland birds are highly dependent on private land for habitat. In the U.S., 85 percent of grasslands are privately owned, and 82 percent of the distribution of 29 obligate grassland species is on private land (North American Bird Conservation Initiative, U.S. Committee 2013). Grassland birds are, therefore, good indicators of areas where human land uses may impact the overall ecological integrity of the landscape. In areas experiencing rapid land conversion, the presence or absence of grassland birds can be used to assess the availability of habitat across the landscape and to set goals for habitat configuration.

Though changes in agricultural landscapes in the Northeast could limit the ability of these grasslands to support grassland bird assemblages (Bennett et al. 2006), grasslands in this region could provide important habitat for these species that are declining throughout their historic range, especially since a higher concentration of breeders in the region are considered eastern subspecies (Wells and Rosenberg 1999). Along the East Coast, grassland birds have been able to use highly artificial grassland habitat, such as hay meadows, uncultivated farmland, and the grassy areas along airport runways (Askins 2002), which can serve as refugia when native grasslands are not available and could be an important factor in these species’ persistence (Herkert, 1994; Vickery et al., 2000).
**Chester County: A Case Study**

Chester County in Southeastern Pennsylvania provides an opportunity to examine the effects of agricultural land use and conservation efforts on grassland bird habitat occupancy. The county lies in the Upland section of the Piedmont Physiographic province, and for much of its Colonial and post-Colonial history, land use has been dominated by rural, agricultural uses. In the 1980s, the county’s economy began diversifying along with its steady population growth, and land uses shifted to residential, commercial and industrial development (Board of County Commissioners 2009). The county recognized the threat of unbounded development on its natural landscapes and agricultural industry and developed its first comprehensive plan, *Landscapes*, in 1996 to guide county land use policies.

In 2009, the county developed its second comprehensive plan, *Landscapes2* (Chester County Planning Commission 2009). The plan’s overall goals included: 1) building and managing a green infrastructure network, and 2) promoting environmentally sustainable farming while maintaining a critical mass of farmland so agriculture and farming support services could continue as the principal industries in the county’s rural areas (Board of County Commissioners 2009). These goals remain with the latest comprehensive plan, *Landscapes3* (Chester County Planning Commission 2018c), produced in 2018, though the emphasis on sustainability is not explicit. The current goals are: 1) preserving the economic viability of farming and the prime agricultural soils in the county by increasing the acreage and clustering of protected active farmland; 2) expanding the protection of natural habitats to ensure the persistence
of wildlife habitat and critical ecological functions; and 3) expanding the network of protected open space by linking natural areas, parks, trails, and farms.

In accordance with these goals, efforts have been underway to slow land conversion and conserve working landscapes and areas of ecological importance. Today about 21 percent of the county’s land cover is comprised of agricultural fields and 14 percent is grasslands (i.e., hayfields, meadows that are no longer cultivated, and serpentine barrens). These grasslands and working landscapes provide bird habitat as evidenced by data from the Breeding Bird Atlas of Pennsylvania (BBA) which documents a diversity of grassland-associated species breeding in Chester County. However, a comparison of first and second BBA periods (1983 – 1989 and 2004 – 2009) for the Piedmont Upland section shows mixed trends in the populations of six grassland species found in Chester County: Bobolink (+43%, p < 0.05), Eastern Kingbird (+2%, p > 0.05), Eastern Meadowlark (-50%, p < 0.001), Grasshopper Sparrow (+31%, p > 0.05), Horned Lark (+316%, p < 0.001), and Savannah Sparrow (+61%, p > 0.05) (Wilson et al. 2012).

Chester County has also experienced habitat degradation, and with rapid and often scattered rural residential development, habitat patches can be isolated within a matrix of development. In addition, loss of habitat within agricultural areas coupled with changes in harvesting schedules for those lands still in agricultural production influence the viability of grassland bird populations in the county. Developing a network of protected

\[ p > 0.05 \text{ indicates a non-significant result.} \]
grassland habitats located among the county’s grasslands and working agricultural lands could reduce the impacts of these land use changes.

Though the grasslands in the county have been ranked according to their conservation values (Natural Lands Trust, 2006), this assessment is limited because ranks are based solely on acreage. If little regard is given to the pattern of preservation—the connectivity and integrity (Merenlender et al. 2004)—the spatial configuration of the resulting network may not be optimal for grassland bird persistence. The conditions in Chester County, therefore, provide an opportunity to implement a science-based conservation plan that would complement preservation goals and help set priorities for strategic land use planning and preservation throughout the county.

**Research Objectives**

The objective of my research is to examine the influence of landscape spatial patterns, preservation status, and zoning on grassland bird habitat occupancy in Chester County. This dissertation tests the hypothesis: Landscape spatial patterns and land use planning outcomes can be used to predict grassland bird habitat occupancy in the agricultural belt of Chester County, Pennsylvania. Using grassland bird data and landscape metrics I provide a landscape scale framework for assessing the grassland reserve network and its ability to support grassland birds.

In Chapter 2, I explore the available literature to examine the following questions: 1) What are the biological implications of human land use patterns? 2) What policy mechanisms are in place to facilitate conservation planning? 3) What strategies can be employed to conduct ecologically responsive land use planning? and 4) What are the
barriers to implementing these strategies? Chapter 3 provides an overview of land use in Chester County, focusing on the factors that shape settlement patterns and land use priorities. Chapter 4 describes the methodology for predicting grassland bird habitat occupancy in a 750 square km area in the southcentral and western part of Chester County. In Chapter 5, the results of grassland bird surveys conducted in Chester County are analyzed, predictive habitat occupancy models are presented and applied to the study area, and a landscape assessment of current and proposed land preservation efforts in the county is presented. The output maps from this chapter can be used to prioritize conservation areas and provide a model for regional conservation planning for grassland birds. Finally, Chapter 6 discusses the implications of this research, looking at how well land use patterns and activities are presently functioning to support grassland birds and how policy and management decisions could improve current outcomes.

This research goes beyond previous efforts which considered only the size of unprotected agricultural lands (Natural Lands Trust 2006), to examine the spatial configuration of habitat patches, proximate land cover types, and the influence of land uses in the surrounding area, which are likely to influence habitat occupancy (Vickery and Herkert, 2001). The goal of this research is to develop a strategic action plan for Chester County based on landscape scale species distribution and habitat needs that will serve as a model for more wide-ranging preservation efforts in the mid-Atlantic region. The results of this research demonstrate the value of combining biological data with landscape metrics to improve conservation efforts while allowing for the economic use of land and limiting disruption to bird populations.
CHAPTER 2 – CONSERVATION PLANNING: POLICY FRAMEWORK AND MOVING BEYOND CURRENT CONVENTIONS

Introduction

The development and expansion of agricultural systems coupled with increasing urbanization, metropolitan sprawl, and wasteful land use patterns have changed the structure of the U.S. landscape (Bender et al. 1998; Daniels 1998; Beatley 2000; Turner et al. 2001). As human populations grow, there will be increasing pressure on scarce natural resources, which will lead to greater environmental degradation and increase the strain on ecosystems, continuing the trend of biodiversity losses (IPBES 2018). Addressing this issue is complicated by planners’ inability to recognize the impact of long-term environmental degradation and planners’ limited understanding of ecosystem processes that pre-dated land conversion for agriculture or other land uses. This generational environmental amnesia (Kahn 1999) and the gradual process of this degradation increases the irreversibility of the threat.

In time frames relative to human society, most ecological systems have changed slowly, and landscape processes have exhibited relative continuity across human generations (Noon and Dale 2002). These landscape processes relate to: 1) the ecological integrity of the landscape—the capacity to support and maintain a balanced, integrated, adaptive biological system with the full range of elements (i.e., genes, species, assemblages) and 2) the processes expected to occur in the natural habitat of a region (e.g., demography, biotic interactions, nutrient cycling, and species-habitat interactions) (Karr 1996). However, increased land development has reduced ecological integrity and increased the rate of landscape change (McPherson 2009). As a result, land use patterns
have fragmented and degraded natural systems, making native species more vulnerable and threatening biological diversity (Forman 2008). With increasing human-domination of the landscape, the ability of many indigenous plant and animal species to survive will be based primarily on the ecological integrity of natural or semi-natural patches in landscapes occupied and managed by humans and the ability of those systems to maintain biotic interactions (Daily et al. 2001; Bhagwat et al. 2005; Fischer et al. 2005; Harvey et al. 2017).

Planning deficiencies related to biodiversity arise from the growing rift between nature’s patterns and processes and current development trends. Humans both influence and are part of ecological systems; therefore, balancing human and ecological needs requires planning for the triple bottom line of economic, social, and environmental sustainability (Daniels 2009). However, there has been a reluctance to acknowledge the economic and environmental cost of uncoordinated and market-driven planning, so the price of development does not reflect its true cost, which can include unintended consequences and negative externalities. Similarly, the initial price of development may not reflect what is most beneficial to a community over time. For example, it has been cheaper to develop greenfields than to concentrate growth by re-using parts of the built environment that have lost their original use (McMahon 2001). But not taking into account the life-cycle cost of development projects perpetuates land use patterns that mask true costs related to efficiency and sustainability. The current compartmentalized planning paradigm can thus obscure far-reaching costs and result in “market failures” (Bator 1958).
Despite this, planning in the United States has traditionally meant planning for development (Wright and Czerniak 2000; Daniels and Lapping 2005), and when sustainability planning is pursued, it does not feature biodiversity goals prominently enough (Miller et al. 2009). As a result, current planning practices will not reduce the overall declines in biodiversity we are experiencing (Doerr et al. 2013) or stem extinction rates, which are now 1,000 times higher than natural background rates of extinction with future rates likely to be 10,000 times higher (De Vos et al. 2015). Creating a landscape mosaic that supports biodiversity will require land use planning that specifically addresses the impacts of land uses on ecological communities and builds ecological concerns into overall planning goals.

In addition, planners will need to plan for the impacts of climate change on ecosystems and wildlife, which could include stress due to temperature rise, changes in precipitation rates and cycles, increased probabilities for extreme disturbances, sea level rise, and competition from invasive species (Daniels 2014). The Intergovernmental Panel on Climate Change (2014) projects global mean temperatures will rise between 1.5 and 4.5 degrees Celsius above pre-industrial levels by the end of the century. This rate of warming is ten times faster than the last period of glaciation, and these changes in temperature will require a faster rate of adaptation or migration than may be possible for many species (Kolbert 2014). Species will need to move upslope or to higher latitudes in an attempt to shift their ranges to cooler climates; in many cases, there may not be room for these shifts (Roman 2011; Groffman et al. 2014). Problems associated with these shifts may be compounded because interspecific differences in response time may disrupt
ecological assemblages, leading to changes in species associations and a restructuring of ecological communities.

With the coming of new temperature regimes and changes in precipitation, climate change will introduce novel ecosystems that have no current analogs—an historical example of this is plant communities from one million years ago are no longer extant and were replaced by ecosystems that did not exist when they were intact (Williams et al. 2007). At the same time, some systems, such as tropical montane systems and polar regions, may disappear altogether by the twenty-first century (Williams et al. 2007; Groffman et al. 2014). Already some wildlife are becoming climate change refugees, pushed to the extremes of their range with nowhere to retreat because the environmental conditions they require are disappearing (Roman 2011). Though land use planning alone cannot provide adequate adaptation to climate change (Daniels 2014), it will be an important element in providing habitat and havens for vulnerable species and stemming biodiversity losses.

One of the major constraining factors in addressing species’ needs now and for the future is the rate of land conversion. Developed or urbanized areas comprise about five percent of the U.S. land surface, with the greatest concentrations of urban land cover in the Midwest, Northeast, and Southeast (Brown et al. 2014). These areas do not include development dispersed among other land uses (e.g., agriculture and forests) and do not reflect the rapid increase in land conversion—the USDA Forest Service (2018) estimates 6,000 acres of open space (i.e., forests and grasslands, farms and ranches, streams and rivers, and parks) are converted to other land uses each day. By 2100, the amount of developed lands at exurban density or greater is projected to expand in the U.S. between
approximately 19 and 23 percent (Bierwagen et al. 2010). This land development will lead to an increase in the amount of impervious surface, which alters the movement of water, energy, and living things (Brown et al. 2014), and to landscape fragmentation, which disrupts ecological processes and species composition (Saunders et al. 1991; Botequilha-Leitão and Ahern 2002; McGarigal et al. 2002).

Relying on federally-protected lands and nature reserves to counter-balance development will be insufficient because privately held lands are too important for conservation—among all federally listed endangered or threatened species 95 percent rely on private lands for some portion of their habitat, and 19 percent exist only on private lands (Wilcove et al. 1996). Moreover, present reserve distribution does not contribute to the full representation of biodiversity (Margules and Pressey 2000; Venter et al. 2014). Frequently, these reserves are relegated to lands that are unproductive or remote, have higher than average elevation, are unfit for human habitation, and/or have limited economic value. Conversely, more productive lands with greater access to water resources tend to be privately held (Scott et al. 2001). As a result, species whose optimal habitats are at lower elevations or are associated with more productive soils are not receiving adequate protection. Thus, within the U.S. reserve network, few of the most productive native habitats are well represented in commercially and culturally valuable landscapes (Franklin 1993; Scott et al. 2001).

There is consensus among planners and landscape ecologists that some type of ecological infrastructure is needed to maintain a landscape that sustains biotic and abiotic resources (Ahern 1995), but to create these conditions, we need greater integration of ecological principles into land use planning (Dale et al. 2000; Groves 2003; Radeloff et
Movement towards greater integration has been hindered by a communication gap between planners and landscape ecologists and conservation biologists. Though ecology and planning are both concerned with the function and optimal use of resources and systems, the transfer of information across disciplines has not resulted in effective problem solving. This has happened because the vast majority of planners have not made biodiversity a focus of their work (Lapping 2006) and have not, traditionally, used knowledge about ecological processes to inform spatial planning (Opdam et al. 2001). At the same time, ecologists have not considered humans as part of the biophysical processes they study (Botequilha Leitão et al. 2006).

Bridging “the rational, compartmentalized reality of science and the emotional, interactive reality of politics” (Rookwood 1995, 380) requires interdisciplinary cooperation (Fry 2001; Hawkins and Selman 2002). Planners need a stronger understanding of the science on which to base their comprehensive plans and area plans, and scientists need to develop operational models that can interact with and influence the planning process (Rookwood 1995; Theobald et al. 2000; Termorshuizen and Opdam 2009). This model for conservation planning would not diverge greatly from best planning practices, but it does call for reorienting the planning profession to operate with an understanding of certain core ecological principles (Benedict and McMahon 2006).

In this chapter, I review the biological implications of human land use patterns and the theoretical basis for conservation planning. Next, I evaluate policy mechanisms for land conservation and related political considerations that influence the planning process. Then I describe a paradigm for conservation planning and how to use the model to make strategic land use decisions for better ecological outcomes. I conclude with a
description of the barriers that could impede progress as we work to create conservation area networks that support biological diversity.

**Land Use Patterns and Biodiversity**

Land use is closely linked to biodiversity. Land use activities that change the structure of the landscape alter the abundance of natural habitats, introduce new land-cover types, and fragment habitat by dividing continuous habitat into smaller patches that are disconnected and unevenly distributed (Forman 1995; Bender et al. 1998; Turner et al. 2001). Changes in land use can also lead to habitat degradation in which the natural vertical structure and horizontal pattern is simplified and natural flows and movements of plants, animals, water, wind, and energy are disrupted (Dramstad et al. 1996). Most native species are poorly adapted to land use changes that interrupt ecosystem functions and processes they depend on to support life cycle needs (McPherson 2009). The loss and degradation of natural habitat is the primary reason for declines in terrestrial biodiversity (Wilson 1988; Wilcove et al. 2000; Fuller et al. 2007).

Island biogeography theory (MacArthur and Wilson 1967) guides our understanding of changes in ecosystem dynamics when land is converted for human use. As habitat patches become isolated in human-dominated landscapes, they follow patterns consistent with actual islands where species diversity is related to patch area and distance from other similar patches—the larger the area, the more species it will contain, and the farther the patch is from other patches, the fewer species will be present. If conditions within a particular habitat patch change, it can affect the biodiversity within that patch (Strange et al. 2006).
Island biogeography has raised the SLOSS (single large or several small) debate among conservation biologists. The question is whether, in practical terms, it is more likely that several small refuges can be created to sustain certain species or whether the retention of a single large area should be the goal to provide adequate space for large iconic species such as mountain lions (*Puma concolor*) and grizzly bears (*Ursus arctos*) (Forman 1995; Quammen 1996). Though there is no prescription for identifying optimal land use patterns and limiting ecological isolation, landscape ecology provides a foundation for evaluating the spatial arrangement of human structures and identifying the potential ecological implications of alternative arrangements (Turner et al. 2001). The factors that most influence the ecology of the landscape are 1) structure—spatial relationships between different land cover types; 2) function—flow and interaction of elements; and 3) change—alteration of structure or function (Forman and Godron 1986).

Assessing the conservation value of the physical environment is based on an understanding of three basic elements of the landscape and how they relate to one another—patches (and their edges), corridors, and matrix structure.

Patches are habitat remnants surrounded by different land cover types. They exhibit a degree of isolation and result from the reduction of a larger vegetation patch, the introduction of a new vegetation type, a disturbance that alters the structure or function of the system, or the preservation of isolated environmental resources (e.g., wetlands in an urban area) (Forman 1995). Habitat patches are part of the landscape mosaic, which is characterized by a diversity of habitat patch types from high- to low-quality, that have resulted from the pattern of development across the landscape (DeStefano 2009).
Edges are the outer section of a patch where the physical conditions differ significantly from the interior core. Edge effects occur where two adjacent landscape elements meet, resulting in species composition and abundance that differs from the patch interior (Forman 1995). Because species perceive and respond to edges differently from humans—what we see as patches are actually on a continuum from the interior core to one patch and then another (Sisk et al. 2002)—edge effects increase as the contrast between patch types increases and where there is a high edge-to-interior ratio (Fahrig and Merriam 1994). Species tend to move away from the edges of their habitat and further into the interior, effectively reducing core habitat area and threatening the persistence of non-edge species (Vickery et al. 1994).

The source-sink model of habitats (Pulliam 1988) and metapopulation theory (Levins 1969, 1970; Hanski 1996; Hanski and Simberloff 1997) provide a framework for understanding species persistence in fragmented landscapes. According to metapopulation theory, there is no single population for a species in a given area; rather, there are clusters of individuals distributed among areas of suitable habitat, and the metapopulation will persist if local extinction rates are balanced with recolonization. Among these individual habitat patches, source areas are ideal habitats that enable species to breed readily and grow in size. When these areas become overcrowded, colonists can be sent out to less suitable sink areas. The colony can survive and even breed in these sinks, but they will need to be subsidized by source areas. The possibility

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4 Along this boundary the influences of the surrounding conditions prevent the development of interior environmental conditions (Forman 1995). With more edge the surrounding conditions exert greater influence on the interior habitat conditions (Lovejoy et al. 1986)
for migration affects local dynamics because individuals could have the opportunity to re-establish a population that no longer persists (Hanski and Simberloff 1997); however, as habitats become more distant, the chance for interaction decreases (DeStefano 2009).

Species’ distribution across the landscape then depends on the abundance and spatial relationship of potential habitats (McIntyre and Hobbs 1999; Fischer et al. 2004). If the structure of the landscape changes too rapidly and exceeds rates of species dispersal, regional populations will collapse, but if a species is able to move around the landscape and integrate needed resources throughout its dispersal area, persistence is likely (Fahrig and Merriam 1994). Based on species use of available habitat, individual habitat patches collectively form a critical threshold across the landscape, a point at which additional loss of habitat produces landscape fragmentation that increases isolation of habitat patches and decreases connectivity (With and Crist 1995). Landscape connectivity is then ultimately defined by functional connectivity—the extent to which elements of the landscape facilitate movement among patches (With 1997; Bestelmeyer et al. 2003). Since many species persist as metapopulations, there must be a degree of connectivity to balance losses among some populations with gains among others. Landscapes that exhibit greater connectivity have been shown to support more native species than those where connections have been lost (Soulé and Terborgh 1999). Conversely, when habitat areas fall below critical thresholds, it affects dispersal success, which can lead to isolation and extinction within patches (With and King 1999; Turner et al. 2001).

Critical thresholds can be crossed abruptly. Therefore, conservation strategies may be more successful and cost-effective if planners can evaluate conditions for which
landscape patterns are significant and identify landscape structure attributes necessary to support species persistence in patchy landscapes before thresholds are crossed (Flather et al. 2002; With 2002). This process is complicated by the fact that there is no definitive threshold for minimum viable population size across a network of habitat patches (Sample and Mossman 1997). While population targets are often set according to scientific conventions or budgetary constraints (Margules and Sarkar 2007), minimally viable populations can be orders of magnitude smaller than populations required for ecologically effective populations that maintain species interactions, genetic diversity, and ecological functions (Soulé et al. 2005). Given these considerations, planners must rely on available data and judgments based on scientific knowledge to maintain habitat connectivity and increase the likelihood that populations are viable.

The fundamental lesson for conservation planners is that interactions of species with their environment are complex. Conservation biology and landscape ecology provide insight into these interactions, especially with regard to the relationship among conservation areas and appropriate boundaries for planning (Groves 2003). Planners who can apply these principles to meet biodiversity goals are more likely to have greater success in determining optimal spatial patterns for species persistence. Historically, planners have not drawn enough from these disciplines (Botequilha Leitão et al. 2006; Nassauer and Opdam 2008; Ahern 2013), but bridging this gap will be essential for long-term biodiversity conservation.

In evaluating habitat needs it is important to keep in mind that different species have different space requirements and varying degrees of tolerance for human proximity (Cypher et al. 2010). For example, the area required for a viable population of mountain lion is much larger than for grey squirrel (Sciurus carolinensis) and requires greater isolation from humans.
Policy Mechanisms for Biodiversity Conservation

Currently, the majority of biodiversity protection in the United States is guided by federal policies. Though conservation planning works best when led by local experts (Margules and Sarkar 2007), responsibility for land stewardship and conservation of bi-ecological resources has been effectively transferred to central governing agencies that are not part of the local community (Gustanski 2000). In fact, most biodiversity protection in the United States has been guided by the federal Endangered Species Act of 1973 (ESA). Though the ESA is one of the most important pieces of environmental legislation to date. Without its protections, Scott et al. (2006) have determined as many as 227 species would have gone extinct; nevertheless, the law has a number of shortcomings.

For one, the ESA is underfunded. In the early 2000s Miller et al. (2002, 167) found funding was less than 20 percent of the amount needed “to get the job done”—though a species may be placed on the federal threatened and endangered list, there is no guarantee of funding to fulfill the law’s mandates. To compound this problem, the ESA’s regulation-driven approach is akin to triage followed by emergency room treatment for those species that qualify. The law has been designed to be reactive, only calling for action when species are in critical condition, and, consequently, when it is more costly and technically more difficult to restore a species (Roman 2011). Furthermore, while the ESA prominently refers to habitat protection, it considers species one by one with “piecemeal and disjointed” planning conducted on a site-by-site basis (Noss et al. 1997, 17). This leads to planning that does not foster an holistic understanding of ecological
communities (Roman 2011) and ignores important species interactions with each other and the physical environment. Concentrating on individual sites ignores the fundamental principle that species widely distributed across their native range are less vulnerable to catastrophe, disturbance, or other negative influences across their entire range (Noss et al. 1997).

In an attempt to address habitat needs, the ESA was modified in 1982 to allow for Habitat Conservation Plans (HCPs), which are designed for ecosystem management and can be applied to multiple species. The HCP is a voluntary contract, usually between a private landowner and the federal government, which identifies habitats that should not be developed, because they are critical to species survival and recovery, and areas where development will not impact vulnerable species. Essentially, landowners can develop part of their land if provisions are made to protect and restore habitat and possibly relocate listed species. In this way, HCPs opened up working landscapes as habitat for endangered species with the possibility of accelerating and expanding recovery (Roman 2011).

Though the approach is more holistic than the ESA, HCPs do not go far enough to protect vulnerable species and their habitats. For one, the “no surprises policy” adopted in 1994 to provide “economic and regulatory certainty regarding the overall cost of species conservation and mitigation” shifts the burden of continued responsibility away from the landowner (Department of the Interior, Fish and Wildlife Service 1998). With this policy, if new circumstances arise after the HCP has been completed, new land restrictions will not be imposed; rather, the burden to manage for these changes lies with the federal government (Camacho et al. 2015). In addition, these agreements are not subjected to scientific peer-review (Kareiva et al. 1999), and, typically, they are “highly collaborative,
minimally regulatory, and only modestly funded” (Mason 2011, 412). In a comprehensive review of HCPs, Kareiva et al. (1999) found that HCPs lack biological understanding of the species covered and plans lack clear provisions for monitoring to quantify changes over time and measure responses to interventions. With limited oversight, there is no way to ensure the HCP is comprehensive enough or will last long enough to provide sufficient habitat protection.

For conservation actions to be successful, they need to be implemented at multiple scales of biological organization, from the species level to ecosystems across the landscape (Groves 2003). Though the scope of the ESA has been extended through HCP provisions, concentrating on endangered species does not ensure the protection of existing ecosystems and more generalized ecological integrity (Linehan et al. 1995), nor does it benefit species that occupy other habitats (Miller et al. 2009). The ESA still only protects a small number of species and leaves each state with a range of species that may be in decline and have little or no protection (Cohn and Lerner 2003). This is problematic because using policy-driven targets that set conservation objectives based on non-biological criteria may not address the level of protection or management needed for particular species (Svancara et al. 2005). Further, because HCPs are often developed for individual properties, most are not integrated within a larger regional plan (Kareiva et al. 1999). The small-scale planning of the HCP may not account for influences on species beyond property boundaries and may not be enough to ensure species persistence over time. This is especially the case for HCPs that apply to small areas and for HCPs that expire within a couple of decades.
In addition to its regulatory policies, the federal government has attempted to provide incentives for habitat preservation and management. The Agricultural Act of 2018 (i.e., the 2018 Farm Bill) provides farmers and ranchers conservation opportunities and technical support for developing and implementing management plans through the U.S. Department of Agriculture Natural Resources Conservation Service (NRCS). Of the $867 billion budget for the Farm Bill, which covers a ten-year period, almost $60 billion has been budgeted for conservation spending (McMinimy 2018). Voluntary programs such as the Environmental Quality Incentives Program (EQIP) and the Conservation Reserve Program (CRP) pay agricultural landowners to implement restoration or conservation plans that maintain or enhance the condition of soil, water, air, and other natural resources.

Specifically, EQIP, which is budgeted for a little over $9 billion over ten years, provides payments for a maximum of a 10-year term for changes to production systems that improve conservation outcomes and for conservation practices to restore, develop, protect, and improve wildlife habitat. EQIP funds can be used to address specific natural resource concerns in priority areas. The 2018 Farm Bill provides a longer-term commitment to conservation practices associated with improving wildlife habitat while acknowledging the necessity of compensation based on the implementation costs and operational risks associated with supporting wildlife (U.S. Congress 2018). Up from five percent in 2014, the 2018 Farm Bill requires at least 10 percent of EQIP funding must be used for practices that benefit wildlife (McMinimy 2018).

In contrast, CRP works by reducing production in certain areas. CRP pays landowners to take environmentally sensitive lands out of commodity production for a
defined period (10 or 15 year terms) and implement approved conservation practices such as maintaining permanent vegetative cover and restoring the hydrology of wetlands (NRCS 2014). The number of acres that can be enrolled in CRP increased from 24 million to 27 million with the 2018 Farm Bill (McMinimy 2018).

Historically, NRCS programs have been successful in keeping agricultural lands from being developed and providing wildlife habitat. Since landowners are compensated for conservation efforts, these programs can increase interest in providing wildlife habitat and improving the ecological value of the land (Parkhurst and Shogren 2003). Nevertheless, it is difficult to measure the long-term benefits from these programs because once the contract on the land ends, the landowner may choose not to re-enroll in the program, so there is a possibility any gains made over the enrollment period could be lost (Parkhurst and Shogren 2003; Fairfax et al. 2005; Dayer et al. 2018). The amount of land enrolled also can limit conservation benefits. Under the 2018 Farm Bill, future CRP outcomes may be limited because the number of acres enrolled in CRP will not reach the previous cap level of 32 million acres which was reduced in the 2014 Farm Bill (Chite 2014). Furthermore, conservation under the Farm Bill may have additional costs—Farm Bill programs can be used to support unsustainable agricultural practices, and under the CRP, landowners may receive payments that are two to three times the value of the land without providing permanent protection (Fairfax et al. 2005).

The 2018 Farm Bill also continues the Agricultural Conservation Easement Program (ACEP), which consolidates earlier programs designed to protect and restore wetlands and to prevent non-agricultural uses on productive farmlands or grasslands (Chite 2014). A major goal of the program is to assist farmers and ranchers operating
working farms to keep their land in agriculture production. To this end, the federal government has increased funding to $450 million over ten years to fund non-federal partners to purchase Agricultural Land Easements (ALE) that protect agricultural uses and conservation values of eligible land and Wetland Reserve Easements (WRE) to restore, protect, and enhance wetlands on eligible land (McMinimy 2018).

With the 2014 Farm Bill, funding for easements shifted. Though ACEP has shown a funding bias in favor of preserving wetlands, which tends to benefit wildlife more than farm and ranchland protections, funding for wetland projects was down. In 2014, of the nearly $330 million in funding that was allocated, 68 percent ($223 million) went to wetland easement projects—down from the annual average of $410 million provided for wetland preservation under the 2008 Farm Bill—and 32 percent ($105 million) went to agricultural land easement projects. In FY2014 and FY2015, an average of 130,000 acres were enrolled in ACEP each year, including 80,000 acres annually of ALEs (60 percent), and 50,000 acres annually of WREs (40 percent) (Commodity Credit Corporation 2016). By comparison, over 200,000 acres on average were enrolled in wetland easements each year under the 2008 Farm Bill (NSAC 2014).

The trend toward enrolling more agricultural lands has continued. In FY2017, almost 300,000 acres were enrolled through ACEP easements. Just over two-thirds of the easements were for agricultural land; while, the other third enrolled wetlands in permanent and 30-year easements (Congressional Research Service 2018a). However, enrollment levels are far below demand. Of the 542 ALE applications, 39 percent were funded, and of the 2,336 WRE applications, 19 percent were enrolled (Congressional Research Service 2018a). Although the 2018 Farm Bill allocates more money for ACEP,
it is still unclear how much land preservation under this program will benefit wildlife in the future. Consequently, the need for non-agricultural acquisitions may increase.

The Land and Water Conservations Fund (LWCF), which is the other major federal funding source for land conservation and the preservation of biological diversity and natural communities, will certainly be part of this equation. The LWCF was established through an Act of Congress in 1964 to provide funding for land purchases (i.e., park lands, wildlife refuges, and recreational areas) and the conservation of forest and other habitat. Under the LWCF the federal government provides 50 percent of a project’s funding and the other half comes from state and local revenue and private donations. All land acquisitions made possible through LWCF funding are open to the public and remain forever available for outdoor recreational use (National Park Service 2011).

Since its establishment, the LWCF had helped to preserve over 7 million acres in 41,000 different park, wildlife habitat, and recreation projects (TPL 2014). Despite its success, the LWCF has not been used to full advantage. The LWCF is funded by dedicated royalty payments derived from oil and gas leases on federal lands; however, these funds can be diverted to cover other federal funding gaps. Since the program’s inception, funding has been unpredictable—over $18 billion of LWCF’s funding has been diverted into general revenues for other purposes, and the LWCF has been fully funded at its $900 million annual cap only once (The Nature Conservancy 2012). Since its inception over 50 years ago, Congress has appropriated less than half of the $40 billion accrued in the fund on conservation efforts (Congressional Research Service 2018b).
In recent years the continuation of the LWCF has been uncertain. In 2015, the LWCF was allowed to sunset at the end of its 50-year authorization. Although Congress later extended its authorization for another three years, that ended on September 30, 2018. With the LWCF funding lapsed, the Senate voted to permanently reauthorize the LWCF on February 12, 2019 as part of sweeping environmental legislation designed to protect large swaths of public lands and wilderness (Eilperin and Grandoni 2019). In a bipartisan vote, the House voted 363 to 62 on February 26, 2019 to reauthorize the LWCF. On March 12, 2019, the president signed S.47, the John D. Dingell Jr. Conservation, Management, and Recreation Act to reauthorize the LWCF along with more than 100 individual bills introduced by 50 Senators and several House members (U.S. Department of the Interior 2019). This was largely seen as an environmental victory because a fully-funded LWCF is an invaluable conservation tool. However, it will not be mandatory to spend all of the funds (Eilperin and Grandoni 2019), and only allocating minimal funding could severely hamper conservation planning efforts.

The North American Wetlands Conservation Act (NAWCA) of 1989 is another land acquisition program that supports activities related to the North American Waterfowl Management Plan, an international agreement among the U.S., Canada, and Mexico to provide for long-term protection of wetlands and associated uplands habitats used by migratory birds and other wildlife (USFWS 2015). NAWCA provides federal matching grants for private-public projects designed to protect, restore, and enhance habitats. Priority areas include coastal wetlands, tidal marshes, floodplain hardwood forests, and freshwater riparian zones. In addition to extending habitat, these projects benefit human populations by providing recreational opportunities, improving local water
quality, increasing flood protection, conserving soil, and creating greater coastal resilience in areas of projected sea level rise (NAWCC 2013). The program has been successful in extending wetland habitat conservation efforts. From September 1990 through March 2014, there were over 2,400 projects that restored or enhanced 27.5 million acres of habitat (USFWS 2015). In January 2019, Congress extended authorization of NAWCA through 2024, setting maximum funding levels at $60 million (Heinrich 2019).

The National Fish and Wildlife Foundation (NFWF), which was created by Congress in 1984, is the nation's largest provider of private conservation grants. The NFWF works with public and private sectors to protect and restore habitats and the species they support and advance sustainable habitat management. NFWF leverages public funding with private capital from corporations and matching funds from private foundations for conservation projects. The foundation is unique in its emphasis on developing collaborative conservation projects with a variety of stakeholders, including government agencies, corporations, nonprofit organizations, private landowners, ranchers and farmers, volunteers, and sporting communities. Funding is awarded to projects that promote science-based conservation and that have quantifiable results (National Fish and Wildlife Foundation 2017). Since its inception, the NFWS has spent more than $5.3 billion to support over 17,250 conservation projects (“About the National Fish and Wildlife Foundation” 2019).

Undeniably, the various federal programs aimed at wildlife protection have been integral to wildlife conservation over the last several decades. However, the factors related to ecosystem decline and biodiversity loss (e.g., rapid urban development and
habitat fragmentation), are not easily controlled at the national level. Though these programs have implications at smaller scales, more action must be taken at the local and regional level where losses are occurring.

**Conservation Planning in the Context of Local Planning**

Since most of the authority for land use decisions is vested at the municipal and county levels, it is the local decisions that have the greatest impact on the natural environment (Endter-Wada et al. 1998; McGinnis et al. 1999). However, local planning offices are most strongly involved in matters of zoning and land use (Jepson 2004a) and have had limited direct involvement in protecting ecological resources (Daniels 2014). In fact, ecological degradation is often hastened by local land use decisions (Noss and Scott 1997; Theobald et al. 2000) because “decisions on land use are many in number and diffuse in space and time” (Theobald et al. 2000, 36). They are often made without an understanding of their cumulative effects (Thompson 2004)—an example of what economist Alfred Kahn (1966) described as the “tyranny of small decisions.”

Essentially, biodiversity conservation has not been a primary concern of local land use planning bodies (Brody 2003; Duerksen and Snyder 2005; Berke 2007; Miller et al. 2009; Stokes et al. 2010). When efforts are made to plan for biodiversity goals, these efforts frequently fall under the umbrella of sustainable development, a large-scale and inclusive framework used to guide planning agendas (Beatley 1995; Berke and Manta-Conroy 2000; Berke 2002; Jepson 2004b), which is often applied through piecemeal
efforts\textsuperscript{6} and typically favors anthropogenic interests in the comprehensive planning process (Berke and Manta-Conroy 2000; Berke 2007).

Meeting biodiversity goals that are not as obviously related to humans (e.g., protecting habitats and preserving ecological integrity) does not feature prominently in traditional sustainability planning paradigms (Miller et al. 2009). Among several studies that surveyed a range of respondents with differing levels of active sustainability planning (Conroy 2006; Jepson 2004a; Saha and Paterson 2008; Miller et al. 2009), none of the most often adopted environmental protection activities was directly related to biodiversity. Though studies cited open space policies that attempted to limit development in certain areas, direct growth away from environmentally sensitive areas, or protect ecosystems and farmland (Jepson 2004a; Saha and Paterson 2008; Miller et al. 2009), open space initiatives do not necessarily benefit wildlife if they do not specifically incorporate habitat protection (Lerner et al. 2007). For open space planning to address conservation goals, planners would have to consider such factors as corresponding human uses (e.g., recreation and management), availability of resources for targeted species, and the influence of the surrounding matrix (Lindenmayer et al. 2008). If the focus is on multi-benefit policies that only provide marginal benefits for wildlife and if preservation measures are based on human-centered decisions, efforts are unlikely to advance biodiversity goals (Miller et al. 2009).

\textsuperscript{6} Sustainability planning generally focuses on à la carte activities related to energy efficiency, air quality, water quality and conservation, energy consumption, solid waste reduction and increasing recycling levels, open space and natural resource conservation, smart growth initiatives, promoting local industries, and social justice and equity initiatives (Jepson 2004a; Conroy 2006; Saha and Paterson 2008).
Despite this lack of attention, effective local planning is essential for meeting conservation goals, and existing regulatory approaches allow for more action than is currently being taken. For most jurisdictions, planning centers around the comprehensive plan, which projects planning needs for a 10 to 20-year horizon. Though this timeframe limits long-term planning capabilities (Daniels 2014), it is the best tool for communities to undertake a methodical assessment of past trends that have led to present conditions and to shape the future around a vision that strikes a balance between the built and natural environment. By attempting to limit human activity that threatens foundational elements, such as “soil, biological production, biological diversity, fresh water, oceans, and air” (Forman 1990, 268) and including biodiversity values into the comprehensive planning process, it may be possible to minimize conflict over future growth and conservation needs (Underwood et al. 2011) and to work towards conservation-driven landscape configurations that best fit with existing patterns or could be combined with other land uses (Opdam et al. 2006).

A key component of the comprehensive plan is the Natural Resources Inventory, which provides a factual base for understanding current environmental conditions (Daniels 2014) and enables a community to evaluate its natural assets using criteria that go beyond economic valuation of resources and development opportunities (Daniels et al. 2007). Developing the Natural Resources Inventory enables planners to identify where plant and animal species are located, especially threatened or vulnerable species. Once species are located, it is possible to assess the quality of these habitats and to identify
primary and secondary conservation areas\(^7\) (PA DCNR et al. 1997). Completing the Natural Resources Inventory is part of the fundamental task of assessing the carrying capacity of an area—the physical limit to the amount of development, pollution, and human, plant and animal population growth that can be sustained before there is a decline in environmental quality (Daniels 2014). For this reason, the Natural Resources Inventory is a crucial element of the comprehensive plan, but it is an element that planners tend to minimize.

Beyond the Natural Resources Inventory, zoning can support or undermine conservation objectives. Though zoning has often been used to chase tax revenue through new development (Daniels and Lapping 2005), zoning should shape patterns of land use across the landscape and provide guidance on how spatial patterns should change over time (Daniels et al. 2007). Zoning that aligns with a strong comprehensive plan can be an important tool for conservation if it is used to minimize habitat loss and fragmentation and promote land uses that are integrated with underlying ecological processes (Pollock 2009; Daniels 2014).

Zoning can then be used to direct growth into the most suitable areas while protecting areas that have significant conservation value. By identifying high priority areas that contain vulnerable species or are vulnerable to conversion, it is possible to determine which areas should not be zoned for development, where growth containment

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\(^7\) Primary conservation areas are the most severely constrained lands where development should be prohibited through zoning. Secondary conservation areas include significant features of the natural or cultural landscape with high conservation value, especially those that are vulnerable or threatened (e.g., ecologically sensitive areas, mature woodlands, core wildlife habitats and travel corridors, riparian areas, groundwater recharge areas, productive agricultural soils, greenways and trails, and historic sites) (PA DCNR et al. 1997).
boundaries could be revised to make them more sensitive to ecological functions, and where to prevent rezoning of high value ecological sites in high growth areas (Gordon et al. 2009). Zoning can be developed to provide increasing protection close to high conservation value areas and lessening protection farther from these sites (Noss, et al. 1997; Milder 2007). Where core habitat areas and movement corridors have been delineated, wildlife habitat overlay zones can be placed over base zones to provide extra protection. In this way, the overlay zone provides opportunities to buffer ecologically significant lands and minimize negative impacts on wildlife (Daniels 2014).

Zoning can also be used to concentrate development and leave undeveloped areas. One option is conservation zoning, which features large minimum lot sizes over a large contiguous area. This type of zoning can be used to maximize efficiency and limit impacts on habitat by minimizing the potential number of houses in the countryside. However, the success of this zoning depends on the minimum lot size requirements. Larger lots can reduce the chance of fragmentation by maintaining contiguous, open blocks of land, but if lots are too small, it can lead to exurban sprawl (Daniels 2014) at the same time that it fragments the landscape, creating parcels too small for non-residential uses (e.g., agriculture or forestry) (Bengston et al. 2004). In addition, except in a few special cases where states intervene to achieve specific goals, most states limit conservation zones because they offer no economic use of the land.8 These restrictions

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8 For example, to improve water quality in the New Jersey Pinelands and the Adirondacks of New York, zoning has been set at one house per 40 acres and one house per 42 acres, respectively, but a conservation zone in Pennsylvania can be no stricter than one house per 10 acres.
limit conservation outcomes and can lead to increasing landscape fragmentation if larger properties are subdivided (Daniels 1998).

Zoning can also call for the reconfiguration of subdivision layouts. The goal for these conservation subdivisions is to cluster housing at higher densities to augment a jurisdiction’s existing green infrastructure network and site conservation lands strategically within the subdivision to maximize the amount of contiguous, unfragmented area (PA DCNR and NLT 2009). However, conservation subdivisions will not automatically offer greater conservation value than dispersed settlements. Lenth et al. (2006) found the ecological characteristics within clustered developments were similar to those of traditional dispersed housing. Undeveloped areas were dominated by non-native vegetation and clustered development did not provide enough undisturbed habitat for development-sensitive species—human commensal species occurred at high densities while development-sensitive species remained uncommon. The ecological value of these conservation subdivisions may, therefore, be limited if land is not managed for specific conservation objectives⁹ or if the protected areas are not selected for their conservation values (Milder 2007). For conservation subdivisions to be an effective tool for conservation planning, they need to include strong ecological guidelines and be integrated into a regional-scale conservation plan (Lenth et al. 2006; Mockrin et al. 2017).

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⁹ Conservation subdivisions are often managed by homeowners’ associations that may lack the knowledge needed to meet conservation goals or their management goals may conflict with natural resource conservation (e.g., managing for aesthetics, privacy, or recreational use) (Austin and Kaplan 2003).
Given the many zoning options, zoning should not be a tool simply for separating land uses and locating new development opportunities. Different zoning regulations can be combined to guide growth and meet conservation objectives at the district level, but zoning will only have limited effectiveness when working to achieve better conservation outcomes at a smaller scale. Another issue to consider is the impact of zoning that introduces development near ecologically sensitive areas. An issue not yet adequately explored in the literature is the introduction of dogs, cats, and kids into natural areas (Daniels 2014). Cats, for example, are notorious hunters and are responsible for the deaths of 2.4 billion birds per year (Sizemore 2019). Dogs and kids have the ability to disrupt and destroy habitats (e.g., dogs chance wildlife and cause disturbance reactions [Sime 1999]; all-terrain vehicles (ATVs) cause habitat destruction [Backcountry Hunters and Anglers 2011]). Zoning will not solve these incompatible co-occurrences.

Subdivision regulations can extend the power of zoning by promoting efficient growth and minimizing environmental impacts at the parcel level (Daniels 2014). Subdivision regulations outline minimum design standards that can reduce the impact of development and regulate developments near sensitive areas to protect natural resources and limit fragmentation. For example, regulations that prohibit the removal of vegetation in core habitats or require a certain percentage of the property to remain vegetated can be used to maintain habitat quality. These standards can call for augmenting the township’s green infrastructure by requiring developers to dedicate a certain percentage of the total development area for parks or recreation or to make payments in lieu of land dedication, so land can be purchased in a priority area identified in the Natural Resources Inventory. Having strong subdivision regulations can be more effective in buffering sensitive areas
than clustering development, which can accelerate fragmentation and undermine conservation objectives (Daniels 2014).

Another planning tool that can have important consequences for wildlife protection is the capital improvements program (CIP) (Daniels 2014). The CIP can influence the location and intensity of development by phasing growth, so the timing of development is linked to the timing of public infrastructure projects needed for development (e.g. sewers, drainage, and major roads) (Bengston et al. 2004). A well-managed CIP can also reduce the need to raise revenue to fund projects, which can weaken a jurisdiction’s ability to follow the comprehensive plan (Daniels et al. 2007). In addition, by setting boundaries around where public services will be provided (e.g., limiting extension of sewer and water lines), the CIP can be linked to urban growth boundaries that concentrate development and keep growth-inducing facilities away from areas of high conservation value. The CIP can also be used to fund targeted green infrastructure investments, such as the purchase or conservation of ecologically important lands (Daniels 2014).

With these planning tools, local governments have the power to avoid wasteful patterns of development that degrade ecosystems and natural capital. But to meet biodiversity conservation goals, planners must use the available planning tools and move beyond a generalized sustainability framework. Positioning biodiversity as a core planning issue requires a commitment to: incorporating biotic data at early stages of planning (Miller et al. 2009; Stokes et al. 2010); using biophysical and socio-cultural information to identify opportunities and constraints for land use across the landscape (Steiner 2000); focusing on ecologically important areas based on information about the
physical conditions needed to support priority ecosystems and vulnerable species (Opdam et al. 2006); considering ecological needs when designating areas to remain undeveloped (Marzluff 2002); and combining growth management tools with land preservation (Daniels and Lapping, 2005). Though the majority of planners do not focus on these issues enough, Lapping (2006) argues the “nitty gritty” of sustainability will be derived from strategies that support and protect biodiversity.

The challenge, however, is that ecosystem boundaries are defined by discontinuities in the landscape that are not related to political jurisdictions (Noss et al. 1997), and administrative entities are not organized around natural landscape features (Mason 2011). When there is collaboration among jurisdictions, it is most often focused on infrastructure planning, with limited effort put toward biodiversity and natural resource management. The result is biodiversity planning rarely extends beyond individual jurisdictions (Miller et al. 2009). This problem is further exacerbated by limited regional planning and a high degree of political fragmentation (Daniels 1998) characterized by disjointed local land use authorities that lack the capacity to address growth appropriately and are hesitant to limit individual landowner’s rights (McLaughlin and Machlis 2008). Local governments have shown a reluctance or even hostility towards collaboration (Daniels 1998), which can lead to plans that are not complementary and may even be contradictory or incompatible, undermining the effectiveness of any conservation planning efforts. This fragmented local government structure is especially problematic in the Northeast, where the small-community focus is a major obstacle to wildlife conservation planning that stretches across jurisdictions.
Privately Negotiated Conservation

Governmental planning agencies have shown limited ability to address land conservation needs. At the same time, rapid population growth and strong market forces that value land only for its development potential have degraded ecological systems and services. In response to this market failure and the need for intervention in the land market, private, non-profit land trusts became a force in land preservation in the 1980s. They emerged at a time when development pressure was mounting and the government was not adequately funding land protection programs such as the LWCF or protecting public lands (Fairfax et al. 2005; McLaughlin and Machlis 2008).

Land trusts, which are non-profit, non-governmental agencies, benefit from the perception that they are free from government control and profit-making influences. They have been able to work more effectively than federal agencies in enlisting private lands for conservation (Fairfax et al. 2005) by engaging in “privately negotiated environmental policy” (Rissman 2011, 170). They establish agreements that promote cooperation and create a model for anticipatory planning that avoids the need for governmental involvement and regulation (Breckenridge 1998). A criticism, however, is land trusts’ transactions are private, and opportunities for public comment are “sacrificed to the momentum of the deal” (Fairfax et al. 2005, 256). Nevertheless, the model has been successful, and together national, state, and local land trusts have permanently protected over 56 million acres nationwide, which is an increase of 9 million acres since 2010 (LTA 2015).

For the majority of land trusts, their highest priority for land preservation is the protection of wildlife habitats and important natural areas (Chang 2010). To achieve
these ends, they take on the role of ecological consultants for local governments, land owners, and developers and act as mediating bodies to achieve conservation goals where government regulatory capacity is limited or fragmented. In addition, they can expand on policies that call for integrated and multi-jurisdictional management of ecosystems and act as oversees, bringing together disparate groups and assembling resources to achieve site-specific conservation goals. Land trusts can, thus, avoid the rigidity and lack of responsiveness that characterizes government activity to offer greater fluidity in achieving conservation goals (Breckenridge 1998).

In working to provide greater protections for natural systems, land trusts engage in both land preservation and conservation. For lands that are ecologically sensitive, especially those areas threatened by encroaching development, a land trust may purchase land outright via fee simple acquisition to maintain total control over the property (Parkhurst and Shogren 2003). Though fee simple land purchases are expensive, they are the strongest form of protection to ensure full preservation and control of the land. However, with the high costs associated with outright purchase, this method can only be used in a limited number of areas (Fishburn et al. 2009) and has not been effective in protecting enough high conservation value areas that are vulnerable to development (McLaughlin 2002). In addition, management costs can be higher because the land trust is responsible for managing the land itself (Fishburn et al. 2009).

Limited funding precludes the opportunity to purchase enough habitat for all species to exist in preserved areas (Beatley 2000; Yuan-Farrell et al. 2005). Therefore, the emergence of a less expensive tool, the conservation easement, has been critical to limiting development. A conservation easement is a voluntary, incentive-based
agreement that removes development rights from private land. These agreements can be used to preserve open space, provide wildlife habitat, buffer core reserves, or limit exposure to human intrusion (Elliman and Howell 2010). Although conservation easements are a lower-cost alternative to preservation, they can cause some tension with local governments. As a non-profit, a land trust is exempt from paying taxes, especially property taxes on land it owns. Sometimes local governments are not in favor of land acquisitions by land trusts because they remove taxable land from the tax rolls.

Nevertheless, lands under conservation easement offer a compromise between biodiversity protection and human land use. Conservation easements gain some protection for lands that remain privately owned, but they are not nature preserves even though they may allow public access (Rissman et al. 2007). Although many elements of biodiversity can tolerate some degree of human land use that results in disturbance or alteration of the landscape (Redford and Richter 1999), the value and effectiveness of a conservation easement depends on how the protected land functions as part of the larger ecosystem and whether “preservation” of a parcel changes the trajectory of land use (Rissman 2011). The goal for enlisting private lands should be to provide “additionality” (McKenney and Kiesecker 2010), positive benefits that would not have occurred without the easement in place. For example, to promote what Rosenzweig (2003) calls “reconciliation ecology,” in which species conservation occurs in the midst of human-dominated landscapes, conservation easements can target working landscapes (e.g., agricultural or forestry lands) to extend core habitats or, at minimum, to buffer important habitat areas (Parkhurst and Shogren 2003).
Unlike land conversion for development, which is irreversible and reduces future options for conservation planning, conservation easements are supposed to maintain continuity of land use over time because they do not allow major physical changes to the land (McLaughlin and Machlis 2008). Still, activities on these lands could have cumulative, nonlinear impacts on ecosystem processes and functions (Maesta et al. 2003; Hansen et al. 2005) because landowners must be able to derive economic benefit from the land and some degree of development may be allowed (i.e., building additional farm structures or increasing building footprints). Along with the benefits of preserving land through easements, it is important to consider the limits of these lands because much of the conservation value of land is related not only to preservation but also to management (Fairfax et al. 2005). For example, working landscapes are less likely to function as core habitat to protect species sensitive to human activity because the needed level of protection in core areas may not be attainable without strict limitations on management practices that are sensitive to conservation goals (Noss and Cooperrider 1994; Sayen 1996; McLaughlin 2002). Not all conflicts can be predicted, but in an attempt to reduce conflicts related to land use activities, permissible uses are delineated in the terms of the easement, and landowners are expected to fulfill “stewardship obligations” (McLaughlin 2002, 468).

Monitoring is a crucial piece of the conservation easement agreement. Since intensive human use could interfere with ecosystem processes, land trusts must have the capacity to monitor and enforce the terms of a conservation easement to ensure land management maintains the conservation value of the land (Merenlender et al. 2004). Studies have shown land trusts have been unable to monitor easements adequately, which
interferes with their capacity to steward the conservation rights they have obtained over time (Fairfax et al. 2005). Given these factors, ineffective monitoring coupled with a reduced commitment to conservation goals on the part of the landowner could limit the conservation gains, the results of which, can be comparable to having actually developed the land (Parkhurst and Shogren 2003).

Other issues can also arise from land trusts’ approach to preservation. In their efforts to plan for open space, some of the country’s larger land trusts have devised methods to work outside of the traditional channels to operate as “de facto” planning organizations (Wright and Czerniak 2000). This role of consultant, however, can work against conservation outcomes. Though the driver for conservation should be based on an understanding of land use and possible use changes at the landscape level, other factors have influenced conservation decisions, including: landowner willingness to sell, goals of local interest groups, property rights and land use policies, and government incentives/priorities, which can all go against scientifically-based prioritization (Newburn et al. 2005).

Land trusts vary greatly in their size, geographic region, capacity to preserve land, and overall effectiveness. The Nature Conservancy is the premier nationwide and international land preservation land trust, that uses a science-based approach to land preservation with a preference for preserving large parcels. But on the other extreme, many of the 1,700 land trusts are small, volunteer-only organizations that have preserved at best only a few thousand acres. They often rely on volunteer staff or individuals without a biology background to evaluate projects’ conservation values and conduct site assessments (Wilson 2011). For many of the smaller land trusts with limited funding,
reserve selection is often guided by opportunity (Merenlender et al. 2004) with a focus on protecting individual parcels (McQueen and McMahon 2003). Because land trusts prioritize voluntary preservation of private lands, the result can be an “incremental and somewhat haphazard system of land protection” (McLaughlin and Machlis 2008, 1567). The challenge for these small land trusts is to ensure the lands they protect do not become “islands” of preservation disconnected from habitats and natural resources across the landscape (Daniels 2014). They also need to ensure they have the capacity to enforce the terms of the land agreement to ensure conservation values do not diminish over time (Fairfax et al. 2005), which can be especially difficult if staffing is limited.

These factors complicate the planning process and can reduce land trusts’ effectiveness in developing a conservation network, which can make it difficult to create blocks of preserved land that would constitute a “critical mass” to support wildlife over the long run (Daniels 2014). With limited funding for land preservation, conservation efforts need to be more strategic. For more effective planning, Amundsen and Culp (2013) recommend land trusts participate more actively in regional planning to add value to local planning efforts and to have a greater influence on long-term conservation efforts, especially if they are able to influence and catalyze conservation outcomes by offering expertise and tools that might not be available through a conventional approach to comprehensive planning. These authors also note that land trusts with strategic plans preserve more land.

Though land trusts’ interventions are not preferable to more effective planning at the landscape and regional scale, they can play an increasingly important role in biodiversity conservation (Groves 2003). The challenge is to prioritize habitat most
vulnerable to conversion or degradation and to develop a robust habitat network comprised of reserves and buffering private lands. For land trusts to fill the conservation gaps created by traditional land use planning practices, they need to focus on strategic planning and maintaining conservation outcomes over time.

**Strategic Conservation Planning**

Conservation land use planning, or what Benedict and McMahon (2006) call green infrastructure planning, places conservation values at the core of land use planning initiatives and seeks to develop an interconnected network of natural ecosystems by planning for land conservation and development simultaneously. The model attempts to move beyond conventional conservation planning, which has focused on opportunity-guided reserve selection (Merenlender et al. 2004) and protecting individual parcels (McQueen and McMahon 2003), to a more integrative framework. To optimize land use, planning emphasis is not on total area preserved, but on improving ecological conditions and supporting biodiversity through an effective and sufficient “emerald network” (Forman 2008). Ultimately, the goal is for landscape patterns to meet two conditions: 1) the spatial pattern of the landscape supports ecological processes required for resilient populations, and 2) changes resulting from development do not threaten long-term species persistence by reducing populations below a critical minimum size (Opdam et al. 2006). In reality, however, green infrastructure planning is often a remedial exercise in which some development has already occurred and natural landscape elements must be restored and reconnected to maximize the ecological services the green infrastructure provides (Tercek and Adams 2013).
Conservation planning is not a new construct; it has evolved over time in response to distinct planning needs (Daniels 2009). Because of this evolution, this planning framework has not been readily defined within planning literature, but it is essential to identify its key features. First, conservation planning is based on limiting wasteful land use to support healthy communities and maintain the long-term sustainability of ecosystem benefits, services, functions, and resources (Zipperer et al. 2000). With the U.S. population of over 320 million people and projected to grow to 400 million by 2051 (U.S. Census Bureau, Population Division 2019), green fields will continue to be developed. The key, however, will be to minimize greenfield development by redefining “developable lands” and revitalizing cities along with inner suburbs (Daniels 2014). This approach complements regional growth management or “smart growth” strategies that allow for development and economic growth while planning for better social and environmental outcomes (Daniels and Lapping 2005).

The idea is to create complementary patterns across the rural-urban gradient. Daniels and Bowers (1997) proposed a model for efficient regional planning that preserves important natural resources and working landscapes. The model calls for implementing urban growth boundaries (UGBs) that separate land slated for growth and development from land designated to remain rural in character and use. Within the UGB, urban services are provided, particularly sewer and water facilities, but these services are not extended beyond the growth boundary. Consequently, the UGB can influence the timing and location of future growth to generate more orderly and efficient land use patterns (Daniels and Bowers 1997) while maintaining a distinct edge or greenbelt between urban and rural lands (Daniels 2014). To reinforce the distinction between the
land within the UGB and the surrounding landscape, zoning outside the boundary must be low-density and restrictive (i.e., agricultural, forestry, or conservation zoning) (Daniels 1998) and implemented in conjunction with land preservation (Daniels and Lapping 2005) that seeks to preserve large contiguous blocks of land and to preserve land along the UGB to create permanent boundaries (Daniels and Bowers 1997).

Duany et al. (2001) proposed a similar idea with the countryside preserve where objective environmental criteria are used to create a rural boundary much like a greenbelt that promotes connectivity among ecologically important areas. Unlike a UGB, the edge is not based on limiting wasteful development; rather, it is designed to delineate the “permanent countryside” and is based on environmental features such as forests, wildlife habitat, waterways, scenic areas, public recreational areas, and agricultural land. Terrain defines the boundary of the countryside preserve, creating a green framework that flows throughout the urban to rural gradient. For planners, employing some combination of the countryside preserve and the UGB would do the most to limit growth in environmentally sensitive areas and direct growth to designated areas.

The more population growth is directed to designated growth areas, the more wildlife habitat can be protected. However, attempts to limit sprawl that do not identify vulnerable ecological communities has the potential to push commercial or residential development into areas of biological significance, so gains are then offset by equivalent losses. Therefore, to maintain ecosystem integrity and concentrate growth away from habitat that would be most negatively impacted, biodiversity goals must be set initially during the strategic phase, then revisited during the rezoning and development phases to address time lags or changes in environmental conditions (Gordon et al. 2009). In this
way, planners are able to consider the cumulative impacts of development to avoid a “death by halves” scenario (Daniels 2014) in which green infrastructure and open space are lost incrementally with successive planning interventions and what is left is fragmented and unusable for area-sensitive species.

Part of this process is balancing the built and natural environment, so land uses do not exceed the carrying capacity of the environmental system. McHarg (1969) argued that by planning according to the physical carrying capacity of a region and individual tracts of land within the region, planners can develop optimal land use patterns given the physical constraints of an area. This model enables planners to take a longer-term view of land use (Hawkins and Selman 2002) and to form a “comprehensive vision” for the future that includes demographic and economic projections (Rookwood 1995). Furthermore, by combining greenbelts with a semi-natural mosaic, it is possible to create a cohesive large-scale ecosystem network that has the potential to increase resilience (Opdam et al. 2006) and reduce the risks associated with climate change (Opdam and Wascher 2004).

Besides the obvious benefit of providing for biological diversity and ecosystem integrity, which contributes to ecosystem services outputs (Daily et al. 1997), a strategic conservation planning approach offers other benefits as well. By moving from site-based planning to planning around key landscape features, there is more flexibility built into the process (Rubino and Hess 2003). Planning at a regional scale may also reduce costs by creating the opportunity to share responsibilities. For example, if a particular ecosystem type is found in a local jurisdiction but there are larger, more biologically valuable examples of that ecosystem elsewhere in the region, a regional planning effort allows that jurisdiction to concentrate on protecting the best remaining examples of other ecological
resources within its boundaries (Noss et al. 1997). In essence, more coordination within a region can lead to clearer guidance, better use of limited local resources, and better land use patterns.

Additionally, being able to plan for preservation and development removes barriers to conservation planning, demonstrating that planning for biodiversity does not need to exclude economic development (Margules and Pressey 2000). Polasky et al. (2005) found the greatest trade-off between biological and economic objectives occurs when trying to make incremental gains beyond the point of optimization. By eliminating those incremental gains, land can be managed for its economic value (i.e., agriculture or forestry) while providing suitable habitat for a larger majority of species. Unlike conservation efforts that are applied reactively and are guided by regulations aimed at conserving species, planning according to least-cost ecological and human scenarios that use land efficiently can create planning alternatives that offer the opportunity to develop proactive plans before biodiversity or individual sites are threatened (Hawkins and Selman 2002). Proactive planning helps to avoid the need for costly restoration and/or mitigation projects after species are threatened (Karr 1990; Scott et al. 2001).

Given the interrelationship between human society and its living space, it does not make sense to consider human land uses separate from ecological processes and functions. Incorporating biological data with land use data enables conservation planners to take a more creative and informed approach to land planning. If this is understood to be a dynamic, iterative process that is revisited over time as new knowledge is acquired or as socio-economic factors change (Margules and Sarkar 2007), an ecoregion can be planned for in a way that enhances its functions and preserves its integrity.
Reserve Selection

Land use regulations on their own are unlikely to produce spatial patterns that maintain biodiversity and wildlife habitat. These regulations are impermanent and subject to change as the political climate changes. As a result, they can be fairly easily modified, particularly if there is fear of violating the Fifth Amendment prohibition on government “taking” private property without just compensation (Daniels 1998). Without permanent protections on undeveloped lands, especially privately-held lands, there is always the risk of future development. Therefore, successful long-term protection of priority areas (i.e., core habitat and movement corridors) is more likely to occur with permanent protections that support strong land use regulations than through zoning alone, but land preservation is a tool planners have not used enough (Daniels and Lapping 2005).

To maximize the benefits of land preservation and avoid non-strategic land-grabs in which open space with low conservation value is preserved, there must be rational predictors for conserving lands (Yuan-Farrell et al. 2005). Meir et al. (2004) have shown that applying predetermined criteria to site selection out-performs ad hoc conservation investment and static comprehensive conservation strategies, especially in highly degraded areas or where uncertainty about future conservation opportunities is high. Because the conservation value of an ad hoc reserve network can be overestimated by the area it occupies, total area preserved should not be confused with total conservation value (Pressey and Cowling 2001; Rissman 2011).

Determining the best approach to developing a conservation network has not been without controversy. There has been ongoing debate regarding the required number and size of patches for biodiversity conservation. This SLOSS debate first arose almost 40
years ago when Diamond (1975), drawing on MacArthur and Wilson’s (1967) theory of island biogeography, asserted a large reserve is better than a small reserve because a large reserve can support more species over time and will have lower extinction rates.

Simberloff and Abele (1976) refuted this oversimplified view of reserve design, stating different species have different requirements, and it is not possible to follow a single conservation regime. Though a large, contiguous area is needed for animals with large home ranges that consist of different habitat types (e.g., landscape species [Sanderson et al. 2002]), small- and medium-sized areas that are closer together can be beneficial to species that do not benefit from a coarse-textured reserve network that only includes larger sites that are spaced far apart (Franklin 1993).

Though the debate has subsided somewhat, it has never been fully resolved. However, what is now widely recognized is there is no simple solution—large and small patches provide different benefits (Forman 1995). Species have different ecological needs and will respond differently to the size, number, and location of patches. For example, with smaller species, a several small approach might work, but large-bodied and wide-ranging predators, such as grizzly bears, require a larger reserve for persistence.

Therefore, each ecosystem must be studied to determine the best design to maintain the highest level of biodiversity possible (Hilty et al. 2006). The amount of protected land required will depend on plan objectives, area requirements of target species, the physical and biotic characteristics of the planning region, and area needed for ecological processes that maintain habitat structure and species composition (Noss 1996). It should be noted, however, a large reserve might offer greater resilience for all species in the face of
climate change if it allows movement to more suitable areas (e.g., wetter or cooler areas) without interruption from unnatural boundaries (Quammen 1996).

Hobbs (2002) stressed many areas are likely to have de facto ecological networks already in place, so it is the planner’s job to recognize those networks and implement strategies to ensure their protection and enhancement. What is needed is an understanding of ecosystem functions and how target habitats and species operate (i.e., how wildlife uses/disperses among habitat patches and the influence of land uses outside of preserved habitats) (Briers 2002). Once these key issues have been identified, it is then possible to develop a landscape-scale plan that assists planners in targeting high biological value lands that are under the greatest threat from anthropogenic activities (Yuan-Farrell et al. 2005). It can also help avoid the creation of isolated reserve areas that have diminishing conservation value as development moves closer to their boundaries (Ewan et al. 2004).

When considering which sites should be added to the reserve network, it is important to consider the biological losses—based on irreplaceability and vulnerability—that would be sustained by not protecting a targeted habitat (Newburn et al. 2005). To do this, planners need to determine what has been achieved through existing reserve networks, evaluate their limitations and deficiencies, and identify new priority areas that enhance complementarity measured by the contribution a single area makes to the conservation network (Margules and Sarkar 2007). Based on those findings, planners can then identify those sites that require immediate protection to prevent biodiversity losses as well as sites that face less urgent threats but are necessary for long-term species and ecosystem persistence (Noss 1996).
Prioritizing land for conservation efficiency is not only based on the ecological benefits and threats; planners must also consider constraints imposed by the cost of protection, available resources, and socio-political factors. Land cost is another consideration in site selection. Parcels with high land quality for development are usually more expensive than parcels that are not as easily developed because the higher development value influences landowner conversion decisions (Newburn et al. 2005). For example, as residential land prices increase, the probability of urban conversion also increases (Bockstael 1996). In some cases, it might be more effective to target areas that are under less development pressure and are, therefore, more affordable, which would enable more land to be protected (Rissman 2011). But it is also essential to assess vulnerability to future development since it may be economically wasteful to protect lands that are unlikely to be converted in the future (Newburn et al. 2005). In addition, it may be ecologically wasteful to use cost as the main factor for determining the location of protected areas. Venter et al. (2014) cautioned expanding protected areas on the cheapest land may restrict the number of threatened species that can be protected.

Although cost is a factor, the spatial configuration of the habitat network should be the primary consideration in conservation efforts (Briers 2002). The planning unit should be the conservation area network, which includes all areas that perform a conservation function, not just those that have been protected (Sarkar 2003). To be sure, the usefulness of the conservation area network is species dependent. For some species the network would act as a refuge of optimal habitat in times of stress. For others it may only be suboptimal habitat. And for others, still, it may represent the only useable habitat remaining (Margules and Sarkar 2007). Nevertheless, in most cases, a generalizable goal
is to strategically restore connectivity throughout the landscape to link core habitat that will enable movement of wildlife among landscape elements (Hilty et al. 2006).

Corridors, which have become increasingly important in the face of habitat loss and isolation (Dramstad et al. 1996), may provide important linkages to connect core habitat areas, enable greater dispersal among metapopulations (Fahrig and Merriam 1985; Anderson and Danielson 1997), and provide for shifting species distribution in response to climate change, especially when connecting heterogeneous habitats (Noss and Cooperrider 1994). Establishing core reserves surrounded by buffer zones managed to protect conservation areas and connected by dispersal corridors (Noss 1992) has become a fundamental element of green infrastructure planning as well as habitat conservation in urbanizing areas (Daniels 2014). However, as the landscape becomes more fragmented, the level of connectivity and the context and influence of the surrounding non-habitat matrix exert greater influence on species composition (Bennett et al. 2006; Botequilha Leitão et al. 2006). In this way, the corridor concept can oversimplify connectivity (Lindenmayer et al. 2008). Since human activity outside of habitat areas influences ecological functions within a patch or corridor, corridors must be considered within the context of the surrounding landscape (Franklin 1993; Noss et al. 1997). Therefore, it is best to look at patterns over the entire landscape, focusing on a gradient, rather than a patch-corridor approach (Lindenmayer et al. 2008).

The conservation portfolio model, which is based on a group of sites that encompass the full range of conservation targets, provides a framework for site selection (Groves 2003; Burgess et al. 2006). This model provides an holistic vision for planning while focusing on parsimoniousness—using the minimum area required to provide
minimum habitat requirements for long-range persistence (Doncaster et al. 1996). Systematic conservation portfolio planning should be led by an assessment of habitat conditions, the degree of existing protection, current and future threats to biodiversity within an ecoregion that will undermine conservation efforts, and priorities for resource allocation based on feasibility and leverage (The Nature Conservancy 2000; Groves 2003). By developing a conservation portfolio, planners can define priority planning units that represent the full distribution and diversity of the included ecosystems (Noss et al. 2002). Undoubtedly, these considerations will require planners to determine how large habitat patches will need to be to meet species requirements (a return to the SLOSS debate).

To achieve this end, landscape metrics can be used to quantify land cover types and gain an understanding of how landscape patterns relate to biological phenomena. Modeling spatial dynamics to identify the connections between natural and cultural variables and the degree to which landscape processes influence interactions between the two (Botequilha-Leitão and Ahern 2002; Botequilhã Leitão et al. 2006; Bennett et al. 2006) provides insight into the influence of landscape characteristics on species distribution and abundance (Johnson and Igl 2001; Bakker et al. 2002; Twedt et al. 2007) and the impact of regional land use patterns (Bennett et al. 2004; Bishop and Myers 2005). Assessments can be based on a set of focal targets or components of biological diversity that are selected using a coarse-filter approach or fine-filter approach (Kiesecker et al. 2009). In addition, data on indicator species guilds, such as avian communities,

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10 An example of a coarse-filter approach would be an ecosystem-level assessment in which habitats are distinguished by distinct vegetation type. In contrast, a fine-filter approach would focus on individual
can be layered in to identify areas of ecological importance based on specialized habitat requirements and sensitivity to environmental change and disturbance (Sundell-Turner and Rodewald 2008).

By combining decision rules with the outcomes of landscape spatial analysis and relative species-specific data analysis, it is possible to create a flexible framework for planning within the conservation area network by ranking sites according to irreplaceability and complementarity (Sundell-Turner and Rodewald 2008). Such a plan enables planners to evaluate the conservation value of individual parcels, so when land becomes available, its value is known (Wilson 2011). These scores can then be updated iteratively as new lands are added to ensure network efficiency and parsimony (Margules and Sarkar 2007).

Planning conducted at the landscape level using this framework can maximize conservation benefits by highlighting areas that are most suitable for potential development and that are also least likely to interfere with critical areas for biodiversity conservation. Recognizing this fundamental premise makes it easier to assess the entire landscape, keeping priority areas that cannot be moved in place, but also determining where target goals can be met elsewhere to avoid conflict with development. Once preservation areas are prioritized for the conservation portfolio, which can be part of the Natural Resources Inventory in a comprehensive plan, steps can be taken to maximize the contribution of areas with ecological value that have not been selected (Kiesecker et al. 2010).

species that have specific habitat or biological requirements, have lost significant habitat, or are sensitive to human disturbance (Kiesecker et al. 2009).
Barriers to Conservation Planning

There is no doubt that more effective conservation planning is needed. Research supports the efficacy of landscape planning models that emphasize ecosystem networks and connectivity and move away from preservation models that create isolated protected areas that cannot sustain long-term biodiversity (DeFries et al. 2005; Newmark 2008). However, planning bodies have been slow to incorporate these findings into the planning process. Moving forward, planners will need to be more involved in reverting some of the ecological damage that has occurred as a result of previous planning initiatives. For conservation planning to hold greater weight within the planning process there are several barriers to overcome.

At present, most planners do not emphasize natural systems (Forman 2008), which leaves a major gap in planning outcomes. Planners need to use the Natural Resources Inventory as a foundational element when developing a comprehensive plan. Planning that does not emphasize the Natural Resources Inventory ignores the natural capital of an area, risks degrading natural resources, and misses the opportunity to put a value on ecosystem services that increase resiliency in the face of increasing environmental challenges (e.g., changing water regimes and climate change) (Tercek and Adams 2013). Once the natural assets of an area have been identified, planners can determine the most effective package of protection and preservation techniques for meeting conservation goals given the existing ecological, economic, and socio-political landscape. But this will require planning that is coordinated beyond local boundaries.

A high degree of political fragmentation among local governments—especially the township and village form of local government in the Northeast—weakens planning
efforts aimed at developing landscape-scale green infrastructure and continuous habitat networks (Benedict and McMahon 2006; Kartz and Casto 2008; Mell 2014). To move beyond fragmented local planning decisions and move towards collective decisions that are more ecologically sensitive, local plans need to fit within the context of a regional plan (Rookwood 1995; Mell 2014), which requires strong regional leadership focused on better spatial management. Planners are often in the best position to stretch their role and act as a catalyst in regional conservation efforts (Beatley 2000; Jepson 2004a), but to create a landscape vision, planners need to be sensitized to the weight of small-scale actions that do not consider landscape-level impacts on natural resources and biologically and ecologically important landscape elements. With a greater emphasis on transdisciplinary knowledge and ecological literacy, planners will be better equipped to develop holistic plans for multifunctional landscapes (Fry 2001; Naveh 2001).

Even with a broader knowledge base and larger-scale cooperation, this type of planning may not happen without the strength of regulation. Aside from the Endangered Species Act or Clean Water Act, ecological principles are rarely attached to mandates. Much of the current conservation planning activity relies on voluntary citizen action, government subsidies and tax incentives, public-private partnerships, conservation easements and special funding (e.g., greenway planning, smart growth initiatives, watershed planning) (Mason 2008). The non-statutory nature of these plans can relegate them to "advisory" plans that are easily ignored (Rookwood 1995), and, depending on the

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11 Reliance on federal level command and control regulations to guide ecological planning can limit outcomes. ESA and CWA regulations can be subject to the political climate, and the president or members of congress can seek to weaken regulations (Davenport and Friedman 2018; Davenport 2018)
political leadership, opportunities can be diminished through lack of funding or easing of regulations. Assigning legal or policy standards to ecological guidelines gives scientific information more weight as a part of the decision-making criteria employed by planning entities (Broberg 2003).

However, regulation may offer better protection at a lower cost because it does not rely on land acquisition to meet conservation goals (Fairfax et al. 2005). Although land preservation is often at the center of conservation initiatives, it is rarely an adequate substitute for strong zoning over large areas. The goal should be to use preservation to protect a critical mass of habitat and working lands that will protect ecosystems and important industries and help to block the expansion of residential and commercial sprawl (Daniels and Lapping 2005). But relying too much on acquisition perpetuates the notion that society must compensate landowners for conservation measures, and without such payments, they can develop and manage their property with impunity, even if this produces greater environmental harm (Fairfax et al. 2005). Achieving conservation goals will require a combination of enlisting voluntary cooperation through financial compensation and using the power of regulation to assure basic levels of environmental stewardship are being implemented.

A regulatory approach that includes incentives and mandates may also be necessary to achieve greater jurisdictional coordination (Wilkinson et al. 2005; Baldwin and Trombulak 2007). A top-down approach, in which mandates are passed down from “higher levels” of government, influences local land use planning (Stokes et al. 2010) and is a driver for initiating conservation action (Miller et al. 2009). Requiring scale-integrated plans from the national to the local level, such that plans are consistent with
each other and ecological principles are integrated into action (Ahern 1995), will not put an end to collaborative goodwill agreements, but it may provide the impetus to enter those types of planning arrangements. Mandates of this kind can help promote conservation by urging action in jurisdictions where communities would not push for them. Nevertheless, such mandates would not be easily established. There would undoubtedly be opposition from those who fear excessive government interference with privately held lands, and adopting regulations of this type may not be possible at a time when private property rights are paramount and there is a push for legislation to reduce state regulatory powers (e.g., Florida, Arizona, Oregon) (Mason 2011).

Another barrier to conservation planning is lack of adequate funding (Saha and Paterson 2008). Planners need better funding mechanisms if they are to initiate more biodiversity conservation activity (Miller et al. 2009; Stokes et al. 2010). Jurisdictions with higher levels of funding are more likely to engage in ecological planning and adopt ordinances that require more conservation actions in exchange for permitting land use proposals (Kartez and Casto 2008; Miller et al. 2009). Differences in the availability of resources also affect which planning tools are used.

Tools such as transfer of development rights and incentive zoning may require specialized knowledge and personnel to implement, which is an added expense for planning departments (Stokes et al. 2010). Other tools like conservation subdivisions may be inherently flawed. For example, if widely used, conservation subdivision can result in clustered sprawl because this type of plan typically does not reduce the number of permitted dwelling units; it simply re-configures the spatial pattern of development (Daniels 1997). It is another form of suburban, automobile-dependent sprawl, and is
hardly a solution at a time when a related major goal is reducing greenhouse gas emissions from transportation. Therefore, it is important to understand the impacts of the tools that are used and how to best use them.

Not having the right tools available for conservation planning will limit the scope of projects, making it more difficult to connect them at a landscape scale, which, once again, reduces returns on biodiversity conservation efforts. The permanent authorization of the LWCF was a victory for conservation, and the 2018 Farm Bill has increased funding for the ACEP, which means there will be more easements on farms and ranchlands, grasslands, and wetlands. On the other hand, with federal leadership focused on rolling back environmental regulations and limiting policies focused on environmental conservation, the federal government cannot be the only source for financing conservation. The greatest hope for funding conservation projects will be to balance federal funding with a combination of state and local government initiatives, private landowner cooperation, and contributions from the non-profit sector (Bendick 2010).

Though lack of funding may be an impediment, access to data should not be. Currently there is insufficient biological data available in a format that is useful to planners (Crist et al. 2000; Theobald et al. 2000; Pierce et al. 2005; Azerrad and Nilon 2006; Miller et al. 2009), and much of the research that is done is not a factor during the planning process. Conservation biologists and landscape ecologists are developing tools to map patterns of biodiversity and monitor threats to biodiversity across regions, but these tools have not been engaged enough in the planning process (Margules and Sarkar 2007). This may be because information is only important or helpful if it "represents a socially constructed and shared understanding” agreed upon by policy actors (Innes 1998,
and biotic data are not easily transferred to the local planning forum (Duerksen et al. 1997). However, there needs to be a process in which stakeholders determine what information means and how it should be used in collective decision-making. To establish scientific credibility in public forums, data must be clear, honest, believable, and useful in the context of the decision-making process (Rejeski 1993). To build consensus around ecological principles, ecological concepts and terms must be translated into the language of public policy (Ewan et al. 2004). Data that translates across disciplines can help to close the gap between planning practitioners and conservation researchers and enable greater collaboration. For example, planning around ecological regions rather than political boundaries would be more effective. New Zealand, which has planned according to watersheds since the early 1990s (Pyle et al. 2001), provides a workable model.

Conversely, if decision-makers are provided with technical information that will allow them to develop a more holistic understanding of planning issues, they must be willing to use it. Kartez and Casto (2008) conducted a study to determine whether access to ecological land conservation data [i.e., Beginning with Habitat (BwH)]\(^{12}\) influenced local habitat conservation decisions. They found open space planning decisions were not highly influenced by BwH. In fact, they concluded "planners may impede local policy action that requires information sharing and communication among diverse stakeholders" (478). While use of datasets, such as BwH and NatureServe\(^{13}\) or locally compiled ecological datasets, is not required, these are valuable data that could help in making

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\(^{12}\) Beginning with Habitat is a single integrated GIS dataset of habitat characteristics that were previously separate and incompatible and an interpretive handbook (Beginning with Habitat Program 2002).

\(^{13}\) NatureServe is an international network of biological inventories (NatureServe 2013).
effective conservation decisions and in ensuring resources are not wasted on preserving areas with lower ecological value. Although planners may fundamentally agree collaboration is important (Stokes et al. 2010), it is not occurring frequently enough. Planners need to be willing to use all available resources even if it requires them to seek non-traditional planning data.

Still, encouraging the use of biotic data may be easier to overcome than psychological inertia. Perhaps the most elusive requirement for engaging in broad-scale ecological planning is building the will to act. Initiating conservation action is typically affected by several interacting factors—information, attitudes, beliefs, concern for the environment, capabilities, and external conditions that facilitate or hinder particular actions—with one variable usually acting as a limiting factor (Stern 2000). As such, planners must understand community values and present plans in terms of shared norms. It is important to assess what type of development a community favors to determine the range of policies and programs that are possible in that environment. Planning efforts can then be positioned based on the community values (Jepson 2004b). For example, green infrastructure planning can be presented as a greenway, which implies access to open space and recreation, or as a wildlife corridor, which implies open space related to biodiversity concerns. The outcomes may be the same, but perception may determine whether a plan can be implemented. Essentially, planners can build support for conservation planning initiatives by emphasizing the connection between quality of life and biodiversity (Daly and Klemens 2005; Balmford and Cowling 2006; Miller 2005) and helping individuals understand the benefits of this holistic, multifunctional planning approach.
Conclusion

It is widely recognized that we are experiencing the sixth mass extinction event in our Earth’s history (Wilson 1999; Novacek and Cleland 2001; Kolbert 2014). To reverse these trends will require a strategic and holistic approach to conservation planning that moves away from opportunistic, piecemeal conservation efforts and fully engages the private sector (Scott et al. 2001). As more land is lost to development, it is increasingly important to focus on the conservation of biodiversity and ecological functions that have been taken for granted or have been undervalued. Since almost every community engages in open space planning, if biodiversity objectives are integrated with a broader approach to open space planning through comprehensive planning and land use regulations, these objectives may be more achievable (Rookwood 1995; Cohn and Lerner 2003; Milder 2007).

This approach is gaining credibility as a fundamental component of planning and has the support of the scientific community, which has provided a range of models that advance our understanding of the relationship between land use and ecological communities and move towards better conservation outcomes. Nevertheless, there is still too much distance between the scientific and planning communities. The result is conservation planning gets lost under the sustainability umbrella, and the passage of policies and programs that focus on conserving biological diversity is impeded. Since most planning happens at the local level where resources for conservation planning are likely to be limited, engaging land trust personnel and other local experts in the planning process and using available biological data could provide a means of expanding
ecological knowledge and filling information gaps that would prohibit stronger conservation planning.

Though there are barriers to ecological planning and lack of political will in many cases, there is a growing understanding of the role biodiversity plays in protecting human life. To overcome inertia, planners need to emphasize the advantages of multifunctional landscapes that link biodiversity benefits to economic and social outcomes (e.g., limiting development that exacerbates environmental problems, keeping diverse agricultural activities on the best soils, and concentrating growth to reduce servicing costs and the need for new infrastructure) (Forman 2008).

This will require an approach that unabashedly plans for the futurescape (Fry 2001)—a landscape vision that transcends current practices and outcomes. It will also require a more proactive approach in which threats are identified and turned into opportunities (e.g., environmental mitigation is an opportunity for pursuing biodiversity objectives; planning for growth provides a means to maintain ecological integrity). As planners succeed in creating multifunctional landscapes that have wide-ranging benefits, they will have examples to tout and learn from to garner support for an integrated approach to planning that makes biodiversity an important element of the cultural landscape.

In the next chapter, I present an overview of the socio-economic and environmental landscape of Chester County. I describe the conservation opportunities within the county and the challenges it faces in planning for growth.
CHAPTER 3 - STUDY AREA: CHESTER COUNTY – AGRICULTURE AMID URBANIZATION

Overview

Chester County lies in the Piedmont Physiographic province, which is characterized by smoothly rolling hills and low to moderate relief with elevations ranging from approximately 20 to 325 meters above sea level (Sevon 2000). About 46 percent of the soils in the county have been classified as prime agricultural soils (Class I and II), as defined by the County of Chester using soil geographic data developed by the National Cooperative Soil Survey (U.S. Department of Agriculture, Natural Resources Conservation Service 1997). The physiography of the county has influenced human settlement patterns and the natural communities that are present (Pennsylvania Natural Heritage Program 2015).

Approximately 32 percent of the landscape is built area, 38 percent agricultural lands, 26 percent forest and woodlands, 2 percent open area and parks, and 2 percent water and wetlands. Historically, Chester County’s economic base was agricultural, but since the 1980s some areas that had once been agricultural strongholds have been converted for residential, commercial, and industrial uses, particularly in the northern and eastern parts of the county (Pennsylvania Natural Heritage Program 2015). From 1980 to 2000, the county underwent a rapid transformation as its population grew from 316,660 (Forstall 1995) to 433,501 (U.S. Census Bureau 2000), a 37 percent increase. Land use trends in Chester County reflect these changes and are similar to urban sprawl patterns in other parts of the country in which development first concentrates along major transportation corridors and spreads out to eventually form residential subdivisions and
scattered houses across the landscape (Botequilha-Leitão and Ahern 2002). In Chester County this pattern of development is most notable along Routes 30 and 202. The pattern of much of the new development in Chester County has been low-density, dispersed settlement, which consumes more land per person than suburban development while reducing the amount of land available for agriculture and wildlife habitat (Daniels 1998).

These growth pressures have been shaped by Chester County’s emergence as a regional job center and by its location within the Philadelphia metropolitan region and the Northeast Megalopolis, which runs from Boston to Washington D.C. and is the most densely populated region of the U.S. (Todorovich and Yoav 2011). With a current population of 519,293 (U.S. Census Bureau, Population Division 2019), which has increased by 20 percent since 2000 (U.S. Census Bureau 2000), Chester County is the fastest growing county in Pennsylvania, and the county’s population is expected to reach 662,000 by 2045 (Chester County Planning Commission 2018c).

Planning Opportunities and Challenges

The Chester County Planning Commission (CCPC) had identified two distinct subareas: 1) growth areas dominated by urban and suburban development and 2) rural resource areas characterized by rural development and agricultural landscapes (see Figure 1). The growth areas have developed along the major roadways (i.e., U.S. Routes 1, 30, 100, 202, and 422) in the eastern and central parts of the county. These areas are the population centers of the county and contain large employment centers, commercial centers, and expansive areas of residential development as well as a full range of public infrastructure services. Most of the county’s non-agricultural jobs are concentrated in a
few areas, with over half of all jobs in seven municipalities. This concentration has led to an imbalance between jobs and housing, pushing housing farther from job centers. Though there are pockets of natural land cover in the growth areas, the majority of the landscape is dominated by development-related land uses. Most of the county’s anticipated growth through 2030 would be best suited for these growth areas. Population growth could easily be accommodated with planning that encourages concentrated new development and redevelopment of areas that are underutilized or have uses that will become obsolete (Chester County Planning Commission 2018c).

Figure 1. Map of Chester County Landscape. Subareas of the Chester County landscape identified by the Chester County Planning Commission (Chester County Planning Commission 2013).
Beyond the growth areas, the rest of the county has been designated as rural resource areas (RRA), much of which are more similar in character to the farming landscapes of neighboring Lancaster and Berks counties than to the rest of the urbanized Philadelphia metropolitan area. These rural areas encompass significant environmental, natural, and agricultural resources, including some of the most productive soils in the nation. Residential development in these areas is mostly concentrated in subdivisions or along roadways, and most development is served by on-lot sewer and water systems. The majority of the RRA is zoned for rural residential development. Some of the RRA has been classified as an agricultural landscape with municipal zoning that provides for agricultural uses at one house per 1 acre to one house per 25 acres, and has a range of active agricultural operations (e.g., field crops, hay production, horse farms, and mushroom production), agricultural security areas, and more than 40,000 acres of land under permanent agricultural conservation easements (Digital First Media 2018). In the agricultural landscape, residential development consists of isolated lots and subdivisions, and non-residential uses are limited.

Within the county, land preservation efforts have focused on conserving ecologically significant areas and the working landscapes that are part of the rural character of the non-urbanized areas of the county. Outside of designated “rural centers,” which have been established to accommodate anticipated growth and development in the rural municipalities, plans for future infrastructure investment within the RRAs are limited, making these areas least suitable for development. The remainder of the RRA is

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14 Among all Pennsylvania counties, Chester County ranks second in the sale of agricultural products (Chester County Agricultural Development Council 2012).
the focal point for county agricultural and municipal open space preservation programs as well as conservation and preservation efforts by non-governmental organizations.

The Chester County Agricultural Land Preservation Board (ALPB) was established to protect the County’s agricultural economy and resources by preserving viable agricultural lands through conservation easements. The Chester County Planning Commissioners have also committed to preserving an integrated network of open spaces, to provide for wildlife habitat, agricultural activities, and parks and recreational facilities. In addition, there are 15 land conservation organizations that hold conservation easements or have preserved land through fee-simple purchases. Together these organizations are leading the preservation efforts in the county.

There are, however, forces at work that are undermining the realization of these goals. Residents of the county say they are most attracted to the “farmland, rural character, and natural beauty” of Chester County (Chester County Planning Commission 2007), but as people seek non-urban residences, these characteristics are compromised. From 1990 to 2000, the amount of new residential land increased by 15,000 square feet for each unit of population change (i.e., 15,000 sq. ft./new resident). Between 2005 and 2015, the amount of developed land increased from 158,470 acres to 171,066 acres. About one quarter of the county (119,000 acres) is in large lots that are unprotected (10 acres or greater) (Chester County Planning Commission 2018c) (see Figure 2). These lots represent the potential for a substantial amount of new development that could increase sprawl and wasteful land use if it is not preserved or new, more restrictive land regulations are not put into place.
There have been some changes in land use patterns in recent years. The rate of development is slowing. From 1995 to 2005, 2,739 acres were developed per year, but from 2005 and 2015, that number was more than halved with an average of 1,143 acres developed each year. Additionally, the ratio of square footage per new resident was reduced to less than 6,000 square feet per person. This was achieved by increasing housing density with more apartments, townhouses, and smaller lots. The result was 100,000 new residents were housed on 10,000 fewer acres than had been projected with the previous comprehensive plan, Landscapes2 (Chester County Planning Commission 2018c).
There was also a concerted effort to preserve lands in the county. Since the original *Landscapes* comprehensive plan was implemented, 129,720 acres were permanently protected for land uses including agriculture, habitat, open space, recreational areas, and federal, state, county and municipal parks (Chester County Planning Commission 2018c) (see Figure 3). These preserved lands represent 27 percent of the county’s total land area. The land easements have been administered by the county’s ALPB and by local and regional land trusts and conservancies. Of the total protected lands, 68,418 acres are in active agricultural use.

**Figure 3. Map of Chester County’s Protected Land.** Chester County’s protected lands as of December 31, 2017 (data source: Chester County Planning Commission 2017).
The slowed pace of development coupled with land preservation has made a continued agricultural economy possible, but challenges remain. Though agriculture is still a large sector of the local economy and there is a considerable amount of open space, the resulting land use patterns have produced agricultural areas that are now highly modified, more fragmented, and subject to development pressure.

In the next chapter I present the methodology for the biologically-based landscape analysis. I describe my field research and the variables tested in models developed to analyze habitat occupancy among grassland birds in Chester County. I also describe the methods for a landscape diagnosis of Chester County.
CHAPTER 4 - METHODOLOGY FOR PREDICTING GRASSLAND BIRD HABITAT OCCUPANCY AND LANDSCAPE DIAGNOSIS

Study Area

To demonstrate how ecological criteria can be part of a larger preservation model, I studied a section of southern Chester County in southeastern Pennsylvania that has been designated as an agricultural and rural landscape (see Figure 4). This area of the county is well-suited for an analysis of the influence of landscape patterns and land use planning on grassland bird habitat occupancy because the landscape is a mosaic of agricultural land uses interspersed with some residential and commercial development.

Figure 4. Map of Study Area in Chester County. 750 square kilometer study area in southern Chester County underlain by 2010 Chester County Orthoimagery (data source: Delaware Valley Regional Planning Commission 2010).
Methods

Focal Species

I chose six focal species to represent the grassland guild: Bobolink (*Dolichonyx oryzivorus*), Eastern Kingbird (*Tyrannus tyrannus*), Eastern Meadowlark (*Sturnella magna*), Grasshopper Sparrow (*Ammodramus savannarum*), Horned Lark (*Eremophila alpestris*), and Savannah Sparrow (*Passerculus sandwichensis*). Grassland bird habitat preferences are diverse, and species often respond to habitat features in varied ways (Herkert 1994). Among the focal species, habitat requirements broadly overlap (Herse et al. 2018), but there is enough of a range in their requirements to represent diversity within the grassland guild. For example, Horned Larks nest in sparsely vegetated or bare ground in open fields (Beason and Franks 1974); Eastern Meadowlarks nest on the ground in fairly dense vegetation (Jaster et al. 2012), and Eastern Kingbirds nest in trees in open fields (Murphy and Pyle 2018). Focusing on these species provides an opportunity to evaluate how a single habitat patch meets the habitat needs of individual species and the guild as a whole and how patches relate to each other across the landscape.

Site Identification

To identify the different land cover types throughout the county, I used the Land Use/Land Cover (LULC) dataset for Chester County, PA (Chester County GIS Department 2005), which is a modified Anderson Level IV Classification System based on true color ortho imagery from 2005 and color infrared imagery from 2002. The Anderson database is a classification system that attempts to provide a standardized and...
systematic approach to the presentation of land use and land cover information at four levels of categorization (Anderson et al. 1976). Level IV classification provides the most detailed unit of categorization and at the smallest scale. The detailed land use and land cover codes allow for finer distinctions among land cover types (e.g., some parcels with agricultural coding would not be considered appropriate habitat for grassland birds). Based on my analysis of the landscape, I developed a simplified coding system using criteria specific to the focal species’ habitat preferences.

For the Chester County dataset, I condensed 138 land use codes into nine land cover types: 1) grass-cropland; 2) shrubland; 3) open, other agricultural land, park/recreational land; 4) woodland; 5) wetlands; 6) water; 7) road-transportation routes; 8) residential; and 9) developed (i.e., commercial, industrial, and institutional uses).

Within my classification system, the grass-cropland category is comprised of managed grasslands—hayfields, alfalfa, and pastures, and row crops—and herbaceous grazing land, which encompasses lands dominated by naturally occurring grasses and forbs or lands that are actively managed to include grasses and forbs (Anderson et al. 1976).

The six focal species breed primarily in hayfields or crop fields (Wilson et al. 2012); therefore, I selected potential habitat only from patches in the first land cover type: grass-cropland. From an ecological perspective, patches represent relatively uniform environmental conditions at a particular scale, and patch boundaries represent a contrast between the internal and external patch characteristics (Kotliar and Wiens 1990).

Individual patch boundaries were defined by evaluating connectivity among individual grass-cropland grid cells. To avoid the inclusion of grass-cropland cells that are connected to a larger patch by small “bottlenecks” (Renfrew and Ribic 2008), I applied...
the four nearest neighbors rule, in which grid cells were considered part of the same patch only if they were immediately bordered by another grass-cropland cell to the right, left, above, or below. Because the focal species have been shown to exhibit area-sensitivity (Vickery et al. 1994; Johnson and Igl 2001; Ribic et al. 2009), potential habitat patches were defined as any individual grass-cropland patch that was a minimum of 10 hectares.

To develop a map of potential survey sites, I concentrated my search area on an approximately 750 square km area in the southcentral and western part of Chester County. I focused on this area for several reasons. Most of the search area is within the RRA boundaries, and about 300 square km have been designated as agricultural landscape. Therefore, most of the search area does not include large-scale development that would make the area less suitable for these bird species. Based on records from the 2004-2009 Breeding Bird Atlas,15 68 percent of the focal species detected in Chester County were located within the search area, and density maps of block and point count surveys showed the highest density of focal species detections were within the search area (see Figure 5). Selecting patches from this search area ensured that survey sites were not dispersed too widely across the landscape to make them incomparable and enabled me to maximize the number of field surveys I could conduct over two field seasons given time constraints. Based on these criteria, I identified 645 potential habitat patches.

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15 2004-2009 Breeding Bird Atlas data include records from block surveys within the three atlas regions in Chester County and point counts that were conducted at randomly selected locations within each atlas region (Wilson et al. 2012).
I used a stratified sample to randomly select 145 agricultural patches from the 645 potential patches. Based on the likelihood that patch size influences patch occupancy, I separated the potential habitat patches into five size classes to ensure survey sites were drawn from a range of patch sizes. There were five size classes: 10 – 34 hectares; 35 – 59 hectares; 60 – 84 hectares; 85 – 109 hectares; and equal to or greater than 110 hectares. Additionally, because the level of preservation may also influence patch occupancy,

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16 Patches with a higher proportion of preserved land may offer benefits that unprotected or minimally protected patches do not offer because preservation limits development and is intended to reduce
potential patches were divided into three levels of preservation based on the percentage of the patch that was preserved through fee simple purchase or a conservation easement: High (≥ 66 percent); Medium (34 to 65 percent); and Low (≤ 33 percent). To achieve balance among the size classes with regard to preservation status, I chose an equal number of patches within each size class and at varying levels of preservation (Table 1).

<table>
<thead>
<tr>
<th>Size Class (ha)</th>
<th>Preservation Level</th>
<th>No. of Patches</th>
<th>No. of Point Counts</th>
<th>Size of Habitat Patch (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>10 - 34</td>
<td>Low</td>
<td>12</td>
<td>12</td>
<td>Min: 10.5 Med: 19.2 Max: 30.0</td>
</tr>
<tr>
<td></td>
<td>Med</td>
<td>5</td>
<td>5</td>
<td>Min: 11.7 Med: 25.1 Max: 28.7</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>12</td>
<td>12</td>
<td>Min: 10.1 Med: 14.8 Max: 33.4</td>
</tr>
<tr>
<td>35 - 59</td>
<td>Low</td>
<td>12</td>
<td>24</td>
<td>Min: 36.0 Med: 44.6 Max: 59.3</td>
</tr>
<tr>
<td></td>
<td>Med</td>
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<td>10</td>
<td>Min: 36.1 Med: 42.9 Max: 54.5</td>
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<tr>
<td></td>
<td>High</td>
<td>12</td>
<td>24</td>
<td>Min: 41.6 Med: 48.1 Max: 58.6</td>
</tr>
<tr>
<td>60 - 84</td>
<td>Low</td>
<td>12</td>
<td>33</td>
<td>Min: 60.5 Med: 79.4 Max: 84.8</td>
</tr>
<tr>
<td></td>
<td>Med</td>
<td>5</td>
<td>15</td>
<td>Min: 61.8 Med: 65.7 Max: 83.5</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>12</td>
<td>36</td>
<td>Min: 60.7 Med: 75.8 Max: 85.0</td>
</tr>
<tr>
<td>85 - 109</td>
<td>Low</td>
<td>12</td>
<td>45</td>
<td>Min: 85.6 Med: 97.4 Max: 110.8</td>
</tr>
<tr>
<td></td>
<td>Med</td>
<td>5</td>
<td>5</td>
<td>Min: 86.4 Med: 89.0 Max: 102.3</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>12</td>
<td>48</td>
<td>Min: 85.0 Med: 91.2 Max: 99.0</td>
</tr>
<tr>
<td>&gt; 110</td>
<td>Low</td>
<td>12</td>
<td>63</td>
<td>Min: 111.5 Med: 141.7 Max: 215.8</td>
</tr>
<tr>
<td></td>
<td>Med</td>
<td>5</td>
<td>41</td>
<td>Min: 140.7 Med: 179.6 Max: 280.4</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>12</td>
<td>84</td>
<td>Min: 117.3 Med: 180.1 Max: 298.5</td>
</tr>
</tbody>
</table>

Patches were selected at random. Within each size and preservation class, patches were assigned values using the RAND function in Microsoft Excel, which gives evenly fragmented (Rissman et al. 2007), and it often involves developing a management plan to protect the conservation values of the land (Wilson 2011).
distributed random numbers between 0 and 1. Patches were ranked according to their assigned random value, and patches with the highest rank were selected first until the requirements of the experimental design were fulfilled. In some cases, patches that had been selected could not be surveyed because I was unable to obtain permission to use the site. When this occurred, I used the rankings to select the patch with the next highest rank. Figure 6 shows the patches I sampled, classified according to the five size classes.

![Figure 6. Map of Sampled Patches](image)

**Figure 6. Map of Sampled Patches.** Sampled patches classified according to area and represented by size class.

**Bird Surveys**

For each patch selected for the survey, I conducted point counts on 100-meter fixed-radius circular plots (Drapeau et al. 1999) to determine the presence or absence of
the six focal species. I conducted one point count per 25 hectares. The number of census plots per site were proportional to the size of the field, with one plot for fields in the 10 – 34 hectare class and up to 11 plots on a site that was greater than 275 hectares. Census plots were located randomly throughout each patch using the Create Random Points function in ArcGIS (ESRI 1999), which provided Universal Transverse Mercator coordinates for each point. The center of the census plot was located at least 50 meters from field edges, and on sites with more than one census plot there was a minimum separation distance of 250 meters between each plot (Vickery et al. 1994).

The surveys were completed between May 21 and July 11 in 2014 and May 16 and July 2 in 2015 to coincide with the focal species’ safe dates for breeding in the study area (Wilson et al. 2012). All point counts were conducted from dawn until 1030 hours on days when conditions were suitable (i.e., no precipitation and no strong winds). Each point count began after a settling period after arrival at the survey point. A point count consisted of standing at a fixed location and recording the presence and approximate distance (i.e., within 50 meters and 50 – 100 meters) of all observed and heard species for a period of 10 minutes. Birds that were flying were counted only if they were using the circular plot (e.g., foraging or displaying) (Johnson and Igl 2001). The census did not include a measure of breeding success (i.e., coding for breeding behavior or nest searching), because the presence of focal species within the census plot, indicated

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17 Safe dates are used to reflect the period in which species that are detected visually or audibly can be safely considered a possible breeding species. These dates are narrower than the species’ breeding season, but they are used to exclude observations of non-breeding birds when compiling breeding records. For the most part, the safe dates for the focal species overlapped, and the surveys fell within the safe dates. However, some observations of the focal species were made outside of the safe dates, but because these surveys were not strictly focused on breeding, I included these observations.
a bird was using the patch, either for nesting or foraging, and, therefore, fulfilling part of the species’ habitat requirements.

After the point count was completed, the census plot was characterized according to the dominant vegetation. Information was recorded related to field type (i.e., hay, row crop, or a mixture), the diversity of plant species, the density of the vegetation, and vegetation height. Environmental conditions—ambient temperature, cloud cover, windspeed, and noise—were also recorded.

A total of 468 census plots were sampled during this study. Rather than surveying a small number of localized patches multiple times, I conducted one survey per census plot to increase coverage of the area and obtain a large sample over the study area (Meentemeyer 1989). However, to ensure the accuracy of my rate of detection, I used a double-sampling approach for a subsample of patches during both field seasons (Bart and Earnst 2002). In 2014, I returned to 8 percent of the census patches. In 2015, I returned to 12 percent of the 2014 census patches and 11 percent of the 2015 census patches. The presence/absence data obtained from the point counts were used along with local and landscape metrics to predict which patches were likely to be occupied by the focal species.

Landscape and Local Variables

Landscape configuration and composition impact ecological processes and organisms independently and interactively. Landscape configuration metrics provide information regarding spatial character and arrangement, position, or orientation of landscape elements; while, landscape composition metrics relate to the variety and
abundance of different land cover types (McGarigal 2015). To calculate spatial extent and configuration (i.e., the spatial character and relationship of individual patches to other patches and patch types) for grass-cropland patches in my study area, I used FRAGSTATS (McGarigal et al., 2012), a computer software program designed to compute landscape metrics using categorical land cover data.

For each grass-cropland patch, 15 landscape metrics were calculated. Together these indices quantify individual patch characteristics and their relationship to other patches and patch types across the study area, including patch size and shape, the degree of contrast among patch types, and aggregation of similar patches. The primary size metrics are measurements of total area (AREA) and the total length of the perimeter (PERIM) including any internal holes in the patch. Perimeter is a representation of patch edges; together these edges help to define overall landscape patterns. In addition, the radius of gyration (GYRATE) is a measurement of the extent of the patch across the landscape and is affected by patch size and the compactness or elongation of a patch. It is, in essence, a measurement of the average distance an organism can travel from a random starting point before crossing over the patch boundary (McGarigal 2015).

Shape metrics are concerned with geometric complexity. The perimeter-area ratio (PARA) is a simple measurement of shape complexity that varies with the size of the patch, so that holding the shape constant but increasing the size of the patch would result in a decrease in the perimeter-area ratio. The shape index (SHAPE) provides a similar measurement, but it standardizes the outcome by comparing the patch shape to a square of the same size to eliminate the size dependency issues related to the perimeter-area ratio. The fractal dimension index (FRAC) is based on the patch perimeter-area
relationship and reflects the degree of complexity of a patch without regard to spatial scale. Patches with simple perimeters approach an index of 1, and patches with more convoluted perimeters approach an index of 2 (McGarigal 2015). The related circumscribing circle index (CIRCLE) is another measurement of patch compactness that compares the area of the patch to the smallest circle that can circumscribe the patch (Baker and Cai 1992). More compact patches approach a reading of 0, and more elongated patches approach a reading of 1. This index is useful in identifying patches that are both elongated and narrow, which indicates there is more edge exposure for organisms. The contiguity index (CONTIG) provides information about the patch shape with regard to the spatial connectedness of the individual cells within a grid-cell patch (LaGro, Jr. 1991).

Core area is the area within a patch from a specified edge buffer. Core area measurements can help in determining edge effects in which different environmental conditions along patch edges lead to avoidance of those areas (Renfrew et al. 2005; Fletcher, Jr. et al. 2007). For this study, core area (CORE) was calculated with a depth-of-edge distance of 50 meters from the patch perimeter (Winter et al. 2000; Bakker et al. 2002; Renfrew et al. 2005; Keyel et al. 2013). Core area can be affected by patch shape and can, thus, have multiple core areas. The number of core areas (NCORE) equals the number of distinct core areas within a patch. The core area index (CAI) quantifies the percentage of the patch that is comprised of core area.

Contrast and aggregation metrics quantify landscape heterogeneity. Contrast reflects the degree of difference between a patch and the surrounding land cover with regard to specific ecological considerations (Kotliar and Wiens 1990). The edge contrast
index (ECON) measures the intensity of contrast between a patch and the adjacent patches. Aggregation metrics are an indication of the spatial distribution of patches. Euclidean nearest neighbor distance (ENN) is a measure of patch isolation. The ENN metric calculates the shortest distance between a patch and its nearest neighbor of the same land cover type. Unlike the ENN, which does not consider the size of neighboring patches of the same type, the proximity index (PROX) distinguishes between sparse distributions of small habitat patches from areas where larger patches are clustered together. Specifically, PROX is concerned with the size and distance of all patches within a defined search radius, and PROX increases as the number of same type patches increases and as those patches become more aggregated. The similarity index (SIMI) was the final landscape metric. SIMI is similar to PROX, but the SIMI is based on the size and proximity of all patches of all land cover types; therefore, focal patches are scored in relation to their similarity to other patches within the given search radius. Specifically, the SIMI distinguishes between configurations in which habitat patches form an assemblage of larger patches that have similar properties from sparsely distributed patches that are small and isolated (McGarigal 2015). For this study, the similarity index provided insight into the level of influence the nine distinct land cover types might have on patch occupancy within a 200 meter search radius (Winter et al. 2006; Renfrew and Ribic 2008).

FRAGSTATS calculates these 15 landscape metrics separately though a number of the measurements are largely redundant and provide alternative formulations of the same information (McGarigal 2015). Therefore, to ensure my models would evaluate independent relationships between habitat occupancy and landscape configuration, I
looked for correlations between the landscape variables. I ran Pearson product-moment correlations between each pair of metrics, which enabled me to remove six variables (see Table 2). For an individual variable to be highly correlated with one or more other variables, the absolute value of the correlation coefficients had to be 0.70 or greater. Using this method, I reduced the number of landscape variables that I would include in the models to a set of 9.

| Acronym | Index (units)                      | Type   | Correlated Variables (> |0.70|)                  | Representative Variable |
|---------|-----------------------------------|--------|--------------------------|--------------------------|--------------------------|
| AREA    | Area (hectares)                   | Local  | CORE (0.960)              | CORE*                    |
|         |                                   |        | PERIM (0.936)             |                          |
|         |                                   |        | GYRATE (0.935)            |                          |
| PERIM   | Perimeter (meters)                | Local  | CORE (0.827)              | CORE                     |
|         |                                   |        | AREA (0.936)              |                          |
|         |                                   |        | GYRATE (0.938)            |                          |
| GYRATE  | Radius of gyration (meters)       | Local  | CORE (0.844)              | CORE                     |
|         |                                   |        | AREA (0.935)              |                          |
|         |                                   |        | PERIM (0.936)             |                          |
| PARA    | Perimeter-area ratio (meters)     | Local  | CAI (-0.784)              | CAI                      |
|         |                                   |        | CONTIG (-0.995)           |                          |
| SHAPE   | Shape index (none)                | Local  | FRAC (0.963)              | SHAPE                    |
| FRAC    | Fractal dimension index (none)    | Local  | SHAPE (0.963)             | SHAPE                    |
| CIRCLE  | Related circumscribing circle     | Local  |                          | CIRCLE                   |
|         | (none)                            |        |                          |                          |
| CONTIG  | Contiguity index (none)           | Local  | CAI (0.784)               | CAI                      |
|         |                                   |        | PARA (-0.995)             |                          |
| CORE    | Core area (hectares)              | Local  | AREA (0.960)              | CORE                     |
|         |                                   |        | PERIM (0.827)             |                          |
|         |                                   |        | GYRATE (0.844)            |                          |
| NCORE   | Number of core areas (none)       | Local  | --                       | NCORE                    |
| CAI     | Core area index (percent)         | Local  | PARA (-0.784)             | CAI                      |
|         |                                   |        | CONTIG (0.784)            |                          |
| PROX    | Proximity index (none)            | Landscape | --                   | PROX                     |
| SIMI    | Similarity index (none)           | Landscape | --                   | SIMI                     |
| ENN     | Euclidean nearest neighbor distance (meters) | Landscape | --                   | ENN                      |
| ECON    | Edge contrast index (percent)     | Landscape | --                   | ECON                     |

*For the Eastern Kingbird model, AREA was used instead of CORE because AREA had a higher correlation with species presence data and resulted in a more parsimonious model.
Within the study area, the majority of the preserved lands have been preserved via conservation easements or agricultural conservation easements. However, placing an easement on a property does not automatically translate into wildlife benefits (Wilson 2011). Therefore, another variable that was tested as a predictive variable was the percentage of that patch that was preserved. This variable was derived by determining the proportion of the patch that had any form of preservation (e.g., easements, fee-simple, parkland, municipal open space). The values ranged from 0 to 1. Table 3 provides summary statistics for the variables used in the models.

### Table 3. Summary Statistics for Study Area

Summary statistics for independent variables assessed within study area.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Mean</th>
<th>Variance</th>
<th>Minimum</th>
<th>Maximum</th>
<th>Std. Deviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Local</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pres%&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.48</td>
<td>0.16</td>
<td>0.00</td>
<td>1.00</td>
<td>0.40</td>
</tr>
<tr>
<td>Veg_Ave&lt;sup&gt;a&lt;/sup&gt;</td>
<td>2.39</td>
<td>1.23</td>
<td>1.00</td>
<td>4.00</td>
<td>1.11</td>
</tr>
<tr>
<td>ALC%Grass&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.73</td>
<td>0.06</td>
<td>0.00</td>
<td>1.00</td>
<td>0.25</td>
</tr>
<tr>
<td>CIRCLE</td>
<td>0.62</td>
<td>0.12</td>
<td>0.32</td>
<td>0.95</td>
<td>0.11</td>
</tr>
<tr>
<td>SHAPE</td>
<td>2.79</td>
<td>0.72</td>
<td>1.24</td>
<td>5.50</td>
<td>0.85</td>
</tr>
<tr>
<td>CORE</td>
<td>32.66</td>
<td>809.90</td>
<td>0.11</td>
<td>158.92</td>
<td>28.46</td>
</tr>
<tr>
<td>NCORE</td>
<td>3.81</td>
<td>13.57</td>
<td>1.00</td>
<td>24.00</td>
<td>3.68</td>
</tr>
<tr>
<td>CAI</td>
<td>35.88</td>
<td>144.41</td>
<td>0.71</td>
<td>59.90</td>
<td>12.02</td>
</tr>
<tr>
<td>Landscape</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PROX&lt;sup&gt;b&lt;/sup&gt;</td>
<td>130.07</td>
<td>15246.58</td>
<td>0.63</td>
<td>662.59</td>
<td>123.48</td>
</tr>
<tr>
<td>SIMI&lt;sup&gt;b&lt;/sup&gt;</td>
<td>1.35</td>
<td>5.03</td>
<td>0.00</td>
<td>12.86</td>
<td>2.24</td>
</tr>
<tr>
<td>ENN</td>
<td>15.06</td>
<td>28.27</td>
<td>12.93</td>
<td>69.64</td>
<td>5.32</td>
</tr>
<tr>
<td>ECON</td>
<td>94.46</td>
<td>45.29</td>
<td>61.30</td>
<td>100.00</td>
<td>6.73</td>
</tr>
</tbody>
</table>

<sup>a</sup> Abbreviations: Pres%, percent of the patch permanently preserved; Veg_Ave, vegetation score based on assessment of dominant vegetation; ALC%Grass, percentage of the patch that is grassland and pasture based on Anderson land Cover data.

<sup>b</sup>Large predictor variable recoded by dividing the number divided by 100 to avoid rounding errors when the variable is included in a model.
Zoning was the final variable tested in the models. White et al. (1997) examined the connection between habitat abundance over time and municipal zoning options that would result in different land development patterns. They showed land development guided by different zoning regulations can affect species persistence. My study area spans 23 municipalities, each with its own zoning ordinance, which opened the possibility that zoning that allows for more residential development or that attempts to limit residential development could affect rates of habitat occupancy. Chester County does not have a comprehensive zoning map; rather, zoning is controlled by the individual municipalities. Therefore, I created a single zoning map for the municipalities in the study area (see Figure 7).

**FIGURE 7.** MAP OF ZONING. 23 municipalities fully or partially contained in the study area, coded according to allowable land uses and minimum area requirements.
Because each municipality has its own zoning codes, I consolidated zoning codes based on allowable land uses and minimum area requirements for development. There were four major zoning categories: agricultural zoning; rural-agriculture zoning, which allows for agricultural or residential development and has different minimum standards for each with the intention of maintaining a more open landscape; cluster-open space zoning, which requires clustering development to maintain a percentage of the gross tract area in open space; and residential and other zoning, which includes commercial, industrial, and institutional development (see Table 4). The intention was to make a distinction between patches located in zones that limit development and protect and promote agricultural land uses and those where zoning allows for more intense, non-agricultural land uses adjacent to the patch. If patches were located in more than one zone, the patch was assigned the zoning code in which the majority of the patch was located.

<table>
<thead>
<tr>
<th>Zone</th>
<th>Type</th>
<th>Land Use</th>
<th>Requirements</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Agriculture</td>
<td>agriculture</td>
<td>agricultural uses only, min. of 1 - 25 acre lot size</td>
</tr>
</tbody>
</table>
| 2    | Rural-Agriculture    | agriculture or residential | agriculture: min. 5 - 20 acre lot size  
|      |                      |                   | rural residential: min. 1 - 5 acre lot size with designated open space requirement, 20 - 60% of gross lot area |
| 3    | Cluster/Open Space   | residential       | low density residential: min. 3 acre (existing) - 10 acre lot size (new construction)  
|      |                      |                   | higher density residential with density multiplier for clustering and designating open space |
| 4    | Other                | varied            | traditional residential, commercial, institutional, or industrial zoning |
Models

I used SPSS software (IBM Corp. 2017b) to construct linear and logistic regression models to evaluate the influence of local and landscape attributes and underlying land use data on the occurrence of grassland birds. I did an analysis of the total number of grassland species per patch and a separate analysis of occurrence for each of the focal species. This approach enabled me to examine factors related to increased patch diversity and to account for the distinct factors that influence patch occupancy among the individual species.

To assess factors related to patch diversity, I constructed a linear regression model using stepwise forward selection (Elliott and Woodward 2007). The response variable in this case was total focal species per patch. The 13 uncorrelated independent variables (r < 0.70) described above were allowed to enter the equation. These variables were tested for normality and skewness. To select the best model, I used Akaike’s information criterion (Akaike 1973) with a second-order correction term (AICc), which is recommended for smaller sample sizes (Burnham and Anderson 2002). The models were compared according to AICc differences because actual AICc values have little meaning on their own. The model that yielded the smallest AICc value was considered the best and most parsimonious model for the set (Burnham and Anderson 2002). If the best model included the variable based on vegetation data from the census plots (Veg_Ave), I also selected a second model that did not include census plot vegetation data using the criteria described above. I developed a second model excluding the census plot data because these data would not be available beyond the surveyed plots, and a model without the local vegetation data could be applied more widely. However, it was important to include
the census plot vegetation variable initially to determine the relative influence of
vegetation on habitat occupancy.

Because the individual focal species have exhibited a range of habitat preferences
(Wilson et al. 2012), I constructed logistic regression models to examine the effects of
local and landscape variables on the occurrence of the individual focal species (Hosmer
et al. 2013). Logistic regression was the most appropriate technique for the dichotomous
response variable where a 1 was given if a species was detected during the point count,
and a 0 was given if it was not. I used stepwise forward selection for the 13 variables that
could be entered into the equation. In addition, I tested for interactions between those
variables. If the best model included an interaction term, I computed a Pearson’s
correlation to assess correlation among the variables in the model. If correlation numbers
were greater than 0.70, I used the mean-centered variables for the variables in the
interaction term to limit the possibility of multicollinearity in the model (Iacobucci et al.
2017). I selected models based on those that produced minimum AICc values with the
fewest number of parameters.

To validate the model and assess its generalizability to a larger data set, I used k-
fold cross validation, a method that divides the dataset into subsets (k = 5); part of the
subset is used to train the model for best fit, and the remaining data are then used to
estimate the accuracy of the model (Arlot and Celisse 2010). For the linear regression
models, final model selection was based on an assessment of the AICc value and the k-
fold adjusted R-square value. For the species-specific logistic regression models, I used
IBM SPSS Modeler (IBM Corp. 2017a) to develop partition models that would provide
k-fold cross validation statistics. Models developed using this method are not assigned an
$R^2$ value. Therefore, I validated the logistic regression models based on k values for the area under the receiver operator curve statistic (AUC), which is equivalent to the Concordance Index C statistic, a measure of goodness of fit for binary outcomes (Hosmer and Lemeshow 2005). Differences in patch characteristics resulting from local and landscape variables likely influence patch occupancy by individual species within avian grassland communities.

**Model Application**

Using probability outcomes from the regression models, I applied the best-fit statistical models for species richness and for occurrence of individual focal species to a 2000-meter buffer around the census patches. I generated spatially explicit predictions for occupancy of grass-cropland patches by the focal species in the buffering landscape. These were used to determine the probability of occurrence in a single patch and to determine the probability of species overlap within the study area.

Using landscape data and county level preservation data (Chester County Planning Commission 2018a, 2018b), I performed a landscape diagnosis to assess current and proposed land preservation efforts in Chester County. A landscape diagnosis builds on the landscape analysis to identify valuable resources and processes as well as landscape dysfunctions and spatial conflicts (Botequilha Leitão et al. 2006). The landscape diagnosis is based on an analysis of spatial patterns quantified by landscape metrics computed for discrete land cover types in FRAGSTATS. The diagnosis also evaluates the impact of county-level initiatives and policies and municipal-level regulations on landscape spatial patterns. I assessed the integrity and connectivity of
grass-cropland patches as well as analyzed spatial patterns within the study area against critical mass thresholds needed to support grassland bird populations and/or to sustain local agricultural activities.

To test the potential impact of new preservation on the focal species, I developed spatially explicit models of the projected preservation landscape. I used a Pearson product-moment correlation coefficient to test for relationships among the occurrence probabilities for individual species within the focal group. There was a high correlation ($r > 0.6$, $p = 0.01$) among two groups of species, so species were separated into four test groups to show how their partitioning of the landscape relates to new preservation (see Table 5). The representative species was chosen on the strength of the regression model determined by the $R^2$ value. The diagnosis was used to determine how well land use patterns and activities are presently functioning to meet agricultural and ecological critical mass goals and to compare future land planning goals to avian probability models.

<table>
<thead>
<tr>
<th>Table 5: Representative Species Groups. Species chosen to represent a subsection of the focal species group in the landscape diagnosis map based on $r$-values $&gt; 0.6$ ($r$-value in parentheses).</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Representative Species</strong></td>
</tr>
<tr>
<td>EAME</td>
</tr>
<tr>
<td>GRSP</td>
</tr>
<tr>
<td>EAKI</td>
</tr>
<tr>
<td>HOLA</td>
</tr>
</tbody>
</table>

In the next chapter, I review the avian census data collected in 2014 and 2015. I present the models developed for the focal species to consider variables that influence
patch diversity and to evaluate the potential differences in how individual species respond to local and landscape factors. I apply the models to the landscape within the study area to generate spatially explicit predictions for habitat occupancy in conjunction with county-wide land preservation maps. I apply the models to the landscape within the study area to evaluate the relationship between planning efforts and habitat occupancy models.
CHAPTER 5 - RESULTS OF SURVEYS, GRASSLAND BIRD HABITAT OCCUPANCY MODELS, AND LANDSCAPE DIAGNOSIS

Survey Outcomes

Among the 145 habitat patches that were censused through 468 points, the number of focal species per patch ranged from 0 to 5 with a mean of 1.6 (see Figure 8). At least one focal species was present in 78 percent of the patches. Of the 113 patches where a focal species was present, 58 percent had more than one focal species. In general, if a species was present at a study site, there was more than one individual. Having more individuals present at a site is an indicator of habitat quality and suggests the patch is suitable for breeding or feeding.

One research objective was to determine the distribution of the focal species throughout the entire survey area. Distribution was measured in terms of how frequently a species was present or absent in habitat patches across the study area. Among the focal species, the most frequently observed species was the Horned Lark present in 45 percent of the patches (n = 65) followed by Eastern Kingbird found in 43 percent of the patches (n = 62). Grasshopper Sparrow was next with observations in 26 percent of the patches (n = 37). The species observed in the fewest patches were Bobolink (17 percent, n = 24), Eastern Meadowlark (16 percent, n = 23) and Savannah Sparrow (14 percent, n = 20). These occurrence statistics may be related to habitat preferences. Horned Larks nest on bare ground and can use open areas in crop fields to establish nests. Grasshopper Sparrows and Eastern Kingbirds can tolerate a range of vegetation structures, with a preference for shorter and less dense grasslands and pastures (Wilson et al. 2012). While
other focal species have more specific habitat requirements, both of these species inhabited a range of habitat types, from tall grasses to row crops.

The composition of the focal species observed shifted between 2014 and 2015. In both years, the most common species were Horned Larks and Eastern Kingbirds. In 2015, Eastern Meadowlarks declined to the least sighted species, and observations of Bobolinks and Grasshopper Sparrows also decreased. The number of Savannah Sparrow observations was the same for both years (see Figure 9).

FIGURE 8. MAP OF SURVEY RESULTS. Results of 2014-15 fixed-radius point counts. The map shows areas where the focal species were present and focal species distribution across the study area. Patches indicated as FS Present had at least one focal species at the time of the survey.
Because some of the focal species are associated with a particular field type, this shift most likely reflects the types of fields censused over the two years, rather than any major population shifts among species. In 2014, hayfields comprised a higher percentage of the sites surveyed, but in 2015, fields with a higher proportion of row crops comprised a higher percentage of sites. Over the two years, species response to vegetation varied. The presence of Bobolinks and Eastern Meadowlarks, which show a preference for mixed grasses and denser vegetation (Martin and Gavin 1995; Jaster et al. 2012), was associated with higher average vegetation scores (VEG_Ave) and higher percentages of the habitat patch that are grassland or pasture (ALC%Grass). Conversely, Horned Larks, which prefer open patches with sparse vegetation (Beason 1995), had lower means for vegetation measurements. Means for the other focal species, which have a wider range of habitat preferences (Wheelwright and Rising 1993; Vickery 1996; Murphy and Pyle 2018), fell between the means for the other species (see Table 6).
Table 6. Habitat Vegetation Scores. Summary statistics for habitat patch vegetation scores when focal species are present.

<table>
<thead>
<tr>
<th>Species</th>
<th>VEG_Ave</th>
<th>±SD</th>
<th>ALC%Grass</th>
<th>±SD</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOBO</td>
<td>3.633</td>
<td>0.588</td>
<td>0.400</td>
<td>0.313</td>
</tr>
<tr>
<td>EAKI</td>
<td>2.492</td>
<td>1.102</td>
<td>0.314</td>
<td>0.247</td>
</tr>
<tr>
<td>EAME</td>
<td>3.865</td>
<td>0.324</td>
<td>0.401</td>
<td>0.293</td>
</tr>
<tr>
<td>GRSP</td>
<td>2.840</td>
<td>1.045</td>
<td>0.284</td>
<td>0.236</td>
</tr>
<tr>
<td>HOLA</td>
<td>1.799</td>
<td>0.668</td>
<td>0.199</td>
<td>0.166</td>
</tr>
<tr>
<td>SAVS</td>
<td>2.470</td>
<td>0.912</td>
<td>0.262</td>
<td>0.163</td>
</tr>
</tbody>
</table>

Figure 10 shows species occurrence and location within the study area between the two survey years. Over the full study, Bobolinks and Eastern Meadowlarks were more clustered within the study area compared to the other four species which were distributed more uniformly.
Patch Diversity Models

I used multiple regression analysis to determine which variables are important to the guild as a whole and to predict which patches have a higher likelihood of occupancy by at least one focal species. The model selection procedure, which was based on the second-order estimate of corrected Akaike’s information criterion values (AICc), resulted in two models that had similar outcomes. Because the “best” model included the variable measuring ground level vegetation gathered during the avian censusing, it was necessary to develop a second model that excluded that data, so the model could be applied to patches that had not been surveyed. Patch diversity was best explained by patch characteristics related to size and vegetation and the landscape context of the patch.

Both models included four variables and explained about 40 percent of the variance (see Table 7). The best model had an adjusted R^2 of 0.417; while, the alternate model had an adjusted R^2 of 0.418. Although this is not an especially robust result, it is comparable to other grassland bird studies (Vickery et al. 1994; Bakker et al. 2002; Hamer et al. 2006). The models shared a number of parameters and were determined to be close in fit based on the difference in their AICc values (ΔAICc). The ΔAICc can be used to rank models. When ΔAICc is between 0 and 2, models can be expected to perform similarly (Burnham and Anderson 2002). With a difference of only 0.133, there is substantial empirical support for both models.
Both models included local and landscape parameters. Three variables were common to both models: core area of the patch (CORE), the percentage of the patch comprised of core area (CAI), and proximity to patches of the same land cover type (PROX). Both models demonstrated the importance of interior habitat area for the species guild. In the best model CORE was the most influential variable (see Figure 11a); whereas, CAI was most important in the alternate model (see Figure 11b). In both models, PROX was the second most important variable. Each model also included a measure of vegetation. In the best model the vegetation metric was the census plot vegetation score (Veg_Ave), which increases as the proportion of grass and hay increases. In the alternate model, the vegetation metric was derived from Anderson land cover data and provided the proportion of the patch that could be categorized as grassland (ALC%Grass). In both models the vegetation metric had a positive regression weight. The measures of vegetation are positively correlated ($r = 0.352$, $p < .000$), which accounts for the similar regression weights between the two models.

### Table 7. Variables in Patch Diversity Models

Table of the significant variables in the multiple linear regression models for patch richness.

<table>
<thead>
<tr>
<th>Dependent Variable</th>
<th>Significant predictor variables*</th>
<th>R² adjust</th>
<th>Kfold R²</th>
<th>ΔAICc</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total No. Focal Species</td>
<td>+ CORE (0.221) + PROX (0.216) + CAI (0.209) + Veg_Ave (0.187)</td>
<td>0.417</td>
<td>0.416</td>
<td></td>
</tr>
<tr>
<td></td>
<td>$p = 0.008$ $p = 0.010$ $p = 0.012$ $p = 0.026$</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>+ CAI (0.243) + PROX (0.249) + CORE (0.197) + ALC%Grass (0.190)</td>
<td>0.418</td>
<td>0.406</td>
<td>0.133</td>
</tr>
<tr>
<td></td>
<td>$p = 0.019$ $p = 0.003$ $p = 0.004$ $p = 0.024$</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*Independent variables ranked according to standardized coefficient; partial correlations in parentheses
Though both models included a measure of vegetation, the greatest determinants of patch diversity were interior patch area and the context of the patch within the landscape. Variables related to the shape and complexity of the patch as well as its similarity to proximate patches were not included in the model. Therefore, if a grass-cropland patch is large enough and not isolated within the landscape, predictions regarding habitat occupancy are not likely to be influenced by other patch characteristics.

Species Occurrence Models

Model selection for individual focal species reflects differences in the way species respond to local and landscape variables. The predictive strength of the models ranged
widely among the species with McFadden $R^2$ values\textsuperscript{18} ranging from 0.603 to 0.155. Nevertheless, the models were similar to or stronger than other predictive models for grassland birds (Vickery et al. 1994; Bakker et al. 2002), and habitat parameters associated with the occurrence of each focal species were largely consistent with known species habitat preferences (Wilson et al. 2012).

When the models were developed, there were several variables that were not significant factors in patch selection for any of the species. CIRCLE, a local variable that measures patch compactness was not included in any of the models. Landscape variables that compare an individual patch only to its nearest neighbors (i.e., ECON and ENN) were also not significant in any of the species models. It is likely these variables are not influential because measures of patch interior area are more important than a measure of patch elongation (CIRCLE). And at the landscape scale, degree of contrast (ECON) and proximity to the nearest grass-cropland patch (ENN) may not measure factors that are important to habitat occupancy, or they may not indicate the degree to which a patch is isolated as well as other metrics. Effectively, the other variables that were significant in the individual species models had more explanatory power.

Similar to the total species models, parameters that provide a measure of interior area were significant predictors for all species except for Eastern Meadowlarks. For Eastern Kingbirds and Savannah Sparrows the core area within a 50-meter buffer (CORE) was positively correlated with patch occupancy. For Bobolinks, Grasshopper

\textsuperscript{18} McFadden $R^2$ is a pseudo $R^2$ that measures the proportional reduction in the -$2$ log-likelihood statistic. It tends to be a little smaller than the Cox-Snell $R^2$, but unlike the Cox-Snell $R^2$, it does not have an upper bound that is less than zero, which can prevent the Cox-Snell $R^2$ from behaving like a linear $R^2$. It also meets most of Kvalseth’s (1985) eight criteria for a good $R^2$ (Allison 2013; Menard 2000).
Sparrows, and Horned Larks, core area was further qualified as the percentage of the patch comprised of core area (CAI), and a larger CAI was associated with a higher probability of occurrence. Other patch specific measurements were significant as well as landscape measurements that penetrated deeper into the surrounding area within a given search radius. For all species except Savannah Sparrow, probability of occurrence was related to a combination of local and landscape attributes (see Table 8).

| Table 8. Variables in Species Models. Significant variables in the “best” and alternate logistic regression models for species occurrence. |
|---|---|---|---|---|
| Species | % Patch Occupancy | Significant Predictors | $R^2_M$ | $\Delta AICc$ |
| Bobolink | 17 | + VEG_Ave (p < 0.001) + CAI (p = 0.007) + PROX (p = 0.018) + CAI (p = 0.007) + ALC%Grass (p = 0.002) + RuralAgZone (p = 0.041) + PROX * Pres% (p = 0.002) | 0.511 | |
| Eastern Kingbird | 42 | + AREA (p < 0.001) - CAI * SIMI (p = 0.015) | 0.154 | |
| Eastern Meadowlark | 15 | + Pres% (p = 0.030) + VEG_Ave (p = 0.001) + PROX (p = 0.008) + Pres% (p = 0.011) + PROX (p = 0.003) + AgZone (p < 0.001) + RuralAgZone (p = 0.011) | 0.603 | 15.427 |
| Grasshopper Sparrow | 26 | + CAI (p < 0.001) + VEG_Ave (p = 0.002) + CAI (p = 0.001) + PROX (p = 0.014) | 0.211 | 4.523 |
| Horned Lark | 44 | + CAI (p = 0.001) - VEG_Ave (p = 0.001) + SHAPE (p = 0.015) - VEG_Ave * SHAPE (p = 0.008) + CAI (p = 0.004) - AgZone (p = 0.005) - ALC%Grass (p = 0.022) + SHAPE (p = 0.042) - RuralAgZone (p < 0.001) | 0.314 | 22.4 |
| Savannah Sparrow | 14 | + CORE (p < 0.001) - CORE * NCORE (p = 0.035) | 0.155 | |

\(^1\) Mean-centered variable

**Bobolink**

Bobolink occurred in 17 percent of the habitat patches surveyed. Though they occurred less frequently than other species, the variance explained by the best model was
relatively large compared to other species models ($R^2_M = 0.511$). Predicted probabilities for Bobolink occurrence are best explained by a combination of patch level and landscape characteristics. In the best model, the most important predictor was census plot vegetation measurements ($\text{VEG\_Ave}$) (OR = 6.618; 95% CI = 2.9 – 15.2). The model suggests if all other variables were held at a fixed value, a census plot with a one unit increase in the vegetation measurement would be over six times more likely to have Bobolinks occupancy. Nevertheless, the strength of this prediction is tempered by the 95 percent confidence interval since a larger confidence interval indicates the odds ratio has a lower level of precision (Szumilas 2010).

Bobolink occurrence was also positively associated with a larger percentage of the patch being comprised of core area (CAI) and greater clustering of habitat patches (PROX). The second model, which loses some explanatory power ($R^2_M = 0.417$, $\Delta$AICc = 14.166), shows a shift in parameters when local vegetation data are removed from the model. CAI became the most important predictor followed by percentage of grassland derived from Anderson land cover data (ALC%Grass). These vegetation data were positively associated with an increase in the probability of occurrence. Zoning was influential in the second Bobolink model such that a patch’s majority location within a rural-agriculture land use zone (RuralAgZone) had a positive effect on occurrence probability. In the second Bobolink model there was a significant interaction ($p < 0.001$) between PROX and the proportion of the patch that has been permanently preserved (Pres%).

To evaluate the interaction, PROX and Pres% were stratified into three levels—below one standard deviation of the mean, within one standard deviation of the mean, and
above one standard deviation of the mean (hereinafter below the mean, at the mean, and above mean respectively)—and compared to the probability of occurrence. At lower levels of PROX, the percentage of preservation had limited effects though levels of preservation above the mean were most associated with an increased probability of occurrence. At higher levels of PROX, low to mean levels of preservation still had limited influence on the probability of occurrence, but when preservation was above the mean, the probability of occurrence increased by more than 10 percent over mean levels of PROX ($\Delta Prob = 0.13$). When PROX was above the mean, mean levels of preservation and greater had the most influence on the probability of occurrence. At mean levels of preservation, the probability of occurrence was over 50 percent higher than low preservation levels ($\Delta Prob = 0.58$), and at preservation levels above the mean, the probability of occurrence was almost 70 percent higher ($\Delta Prob = 0.69$). This interaction is indicative of landscape level influences on local patches. A patch that is preserved is more likely to have Bobolinks if it is closer to other similar patches. On its own, the impacts of higher levels of preservation are limited (see Figure 12).
Eastern Meadowlark

The models for Eastern Meadowlark occurrence are similar to the Bobolink models. The best model ($R^2_M = 0.603$) included three variables. As with the best Bobolink model, this model included census plot vegetation measurements (VEG_Ave) and PROX. The coefficients for both variables were positive. Rather than a measurement of interior area, for this model the other predictor was Pres%. Though preservation was a significant predictor ($p = 0.03, 95\% \text{ CI} = 1.4 - 803.5$). When ground level vegetation was removed from the possible predictive variables, the resulting model lost predictive power ($R^2_M =0.496, \Delta AICc = 15.427$). The second model still included PROX and Pres% ($p = 0.011, 95\% \text{ CI} = 2.30 - 559.6$). The other significant predictor variable was zoning. Patch location within agricultural (AgZone) and rural-agriculture zoning districts
(RuralAgZone) was positively associated with species occurrence (OR = 33.164, 95% CI = 5.5 – 199.7 and OR = 7.393, 95% CI = 1.6 – 34.4, respectively). The model suggests if a patch is in an agriculture or rural-agriculture zoning district, which protects agricultural land and permits only low development density, it is more likely to have Eastern Meadowlarks. These results must be considered along with the confidence intervals. The wide intervals indicate the small sample size affected the model’s precision in predicting outcomes (Nakagawa and Cuthill 2007).

**Horned Lark**

Landscape scale variables had limited influence on predicting Horned Lark habitat occupancy. The best predictors for Horned Lark occurrence are local variables. Though Horned Larks had the highest rate of occurrence at 44 percent, the best model only accounted for a third of the variance ($R^2_M = 0.314$). The CAI was a significant predictor of occurrence (OR = 1.062, 95% CI = 1.0 – 1.1) as well as the shape index (SHAPE), a measure of the complexity of the patch (OR = 2.060, 95% CI = 1.2 – 3.7). VEG_Ave was a significant predictor ($p < 0.001$) with a narrow confidence interval (95% CI = 0.1 – 0.4), but it was negatively associated with Horned Lark occurrence because the patchy, open ground that occurs with row crops and is preferred by the species received the lowest values in the vegetation index. The interaction between SHAPE and VEG_Ave was also significant.

As with the Bobolink model, I examined the probability of occurrence when shape and the vegetation index were stratified into three levels— below the mean, at the mean, and above the mean. Below the mean for percentage of the patch comprised of
grass (ALC%Grass), the probability of occurrence was higher at all levels of SHAPE, from compact to more complex, with the higher probabilities among more complex patches (see Figure 13). As the vegetation score increased, probabilities of occurrence decreased at all levels of SHAPE. When ALC%Grass is above the mean, elevated patch complexity is associated with lower occupancy predictions.

The second model, though not as strong as the best model ($R^2_M = 0.212$, $\Delta AICc = 22.4$), shared the CAI and SHAPE parameters. Without the patch level vegetation variable, ALC%Grass, AgZone, and RuralAgZone became significant predictors of Horned Lark occupancy. An odds ratio of less than one describes a negative relationship between variables. With a unit increase in ALC%Grass ($OR = 0.094$, 95% CI = 0.01 –
0.7), the odds of Horned Lark occurrence decreases. Similarly, if the patch was in an AgZone, the odds of occurrence decreased (OR = 0.116, 95% CI = 0.02 – 0.5), and if it were in a RuralAgZone, the odds of occurrence were even less (OR = 0.090, 95% CI = 0.02 – 0.3). For each of the zoning variables, confidence intervals were narrow, indicating a high level of confidence in the relationship between zoning and occupancy. Within the constraints of this analysis, Horned Lark occupancy prediction is driven almost entirely by patch level parameters.

*Other Focal Species*

The models for the final three focal species have limited predictive power and do not follow similar patterns. In the best Grasshopper Sparrow model (R²_M = 0.211), VEG_Ave is the strongest predictor of occurrence (OR = 1.909, 95% CI = 1.3 – 2.8). The second parameter in the model is CAI, which is also a strong predictor (OR = 1.117, 95% CI = 1.1 – 1.2). Though Grasshopper Sparrows occupied only 26 percent of the patches surveyed, the narrow confidence intervals indicate the sample size was large enough to determine the strongest predictors among the parameters tested. CAI is also a strong predictor in the second model (OR = 1.087, 95% CI =1.0 – 1.1). In this model, the vegetation variable is replaced with only one other parameter, PROX, which is significant (p = 0.014), but with an odds ratio close to one (OR = 1.004) the odds of occurrence and non-occurrence are equally likely. Among the species models, the two models generated for Grasshopper Sparrow have the closest R²_M value (Δ R²_M = 0.028) and the closest AICc values (ΔAICc = 4.523). This indicates the second model for Grasshopper Sparrow is the best “alternate” model.
For Savannah Sparrows and Eastern Kingbirds, there was only one model each; however, each of these models accounted for only about 15 percent of the variance ($R^2_M = 0.155$ and $R^2_M = 0.154$, respectively). Since Savannah Sparrows were detected in only 14 percent of the habitat patches, the limited predictive power of the model is to be expected. However, it is noteworthy that the Eastern Kingbird model is relatively weak because this species had a 42 percent occurrence rate. This indicates there were other factors not tested in the model that influence Eastern Kingbird habitat occupancy. Both models are comprised of local variables, but the model for Eastern Kingbird does include an interaction with a landscape variable.

For Eastern Kingbirds the strongest predictor of occurrence was total patch area (AREA), ($OR = 1.021$, 95% CI = 1.012 – 1.030). AREA was used instead of CORE because the AREA variable created the most parsimonious model with the lowest AICc. There was also an interaction between the CAI and SIMI, which is a landscape variable measuring patch similarity to other land cover types within a predetermined search radius. Measurements for SIMI fell within one standard deviation of the mean or were above that; none was below. Therefore, SIMI scores were categorized as at the mean or above the mean. Scores for CAI were separated into three groups based on being within one standard deviation from the mean or above or below that threshold. Viewing the groups in this way shows CAI at the mean or below the mean is negatively associated with the probability of occurrence as SIMI increases (see Figure 14). The decrease in the influence of CAI is similar for these two groups. However, when CAI is above the mean, it exerts greater influence on occurrence probability when SIMI in centered around the mean, but its influence is less than the other two groups when SIMI is above the mean.
This indicates if a patch is more like other land cover types, the size of the interior area of the patch becomes less important.

Within the scope of the available data, occurrence predictions for Savannah Sparrows were entirely tied to core area measurements. Increases in CORE were associated with an increased probability of occurrence though this influence was not particularly strong (OR = 1.037, 95% CI = 1.0 – 1.1). There is also an interaction between CORE and the number of core areas within the patch (NCORE) (i.e., the number of distinct areas that are more than 50 meters from the patch edge). All CORE and NCORE measurements were within one standard deviation of the mean or greater. Consequently, the interaction was examined only for the two groups (see Figure 15). As
CORE got larger, the probability of occurrence increased when NCORE was at the mean. If NCORE was above the mean, the probability of occurrence decreased as CORE got larger. Though the core area was increasing, distinctions between the separate core areas influenced the suitability of the patch.

![Graph of Savannah Sparrow Model Interaction](image)

**Figure 15. Graph of Savannah Sparrow Model Interaction.** Graph of the interaction in the Savannah Sparrow model showing the change in the expected probability of occurrence by core area and the number of core areas. Vertical bars indicate 95% confidence intervals for predicted probabilities.

**Model Validation**

I validated the species occurrence models using k-fold cross validation ($k = 5$). For the multiple linear regression models, I randomized the data and divided it into 5 folds. Using the first four folds, I created a training set on 80 percent of the data and used the final fold to create a test set using the remaining 20 percent of the data. The test set returned a standard error of estimate. I repeated this procedure on the remaining four
folds, having each enter the model as a test set. The procedure yielded a CVerror of 0.906 for the training set for the best total species model, which was smaller than the mean squared prediction error for the entire model, SE = 0.919. For the second model, the CVerror was 0.879. There was a relatively small difference (ΔCVerror = 0.033) between the full model and the validation model. The results of the cross validation indicate the models can be applied to a different data set in the study set with similar results.

For the multiple logistic regression models, I compared the Concordance Index C statistic from the full model to the average area under the receiver operator curve (AUC) generated by training and testing the models using 5 subsets (Table 9). In each case the AUC generated for the validation model was within ± 0.03 of the full model. The validation statistic was meant to assess how accurately the models will perform if applied to a new data set in the study area, not to assess the strength of the model. The C-statistics for the full models range widely among the different species models, and those with lower C-statistics may not provide high prediction probabilities.

Nevertheless, the k-fold cross validation shows there are relatively small differences between the full model and the training models, and applying the models to a different data set within the study area will yield similar results overall.
TABLE 9. CROSS VALIDATION RESULTS. Cross validation statistics for individual species logistic regression models.

<table>
<thead>
<tr>
<th>Species Model</th>
<th>C-Statistic</th>
<th>Validation Mean AUC</th>
<th>ΔAUC</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOBO</td>
<td>0.940*</td>
<td>0.9304</td>
<td>0.010</td>
</tr>
<tr>
<td></td>
<td>0.896</td>
<td>0.8736</td>
<td>0.022</td>
</tr>
<tr>
<td>EAKI</td>
<td>0.758</td>
<td>0.752</td>
<td>-0.006</td>
</tr>
<tr>
<td>EAME</td>
<td>0.966*</td>
<td>0.967</td>
<td>-0.001¹</td>
</tr>
<tr>
<td></td>
<td>0.940</td>
<td>0.9412</td>
<td>-0.001</td>
</tr>
<tr>
<td>GRSP</td>
<td>0.804*</td>
<td>0.7946</td>
<td>0.009</td>
</tr>
<tr>
<td></td>
<td>0.776</td>
<td>0.748</td>
<td>0.028²</td>
</tr>
<tr>
<td>HOLA</td>
<td>0.855*</td>
<td>0.8814</td>
<td>-0.026</td>
</tr>
<tr>
<td></td>
<td>0.788</td>
<td>0.7934</td>
<td>-0.005</td>
</tr>
<tr>
<td>SAVS</td>
<td>0.736</td>
<td>0.722</td>
<td>0.014</td>
</tr>
</tbody>
</table>

* Best model, ¹ minimum ΔAUC, ² maximum ΔAUC

Model Application

I applied the best-fit statistical models that did not include patch level vegetation data to the landscape within a 2000-meter buffer of the patches in the survey. Within the buffer zone there were 570 patches larger than 10 hectares, 425 of which had not been surveyed as part of this research. The models predict which grass-cropland patches in the study area are likely to be occupied by the focal species.

The species richness model provides a spatially explicit projection of patches likely to be occupied by at least one focal species (see Figure 16). Within the study area, 46 percent of the grass-cropland patches were predicted to have at least one focal species.
Of those patches, 91 percent were outside of the growth areas designated by Chester County, and for 33 percent some portion of the patch was permanently preserved.

For the individual species models, predictions were based on the probability of occurrence of each of the six focal species. The predictions showed differences among the species’ use of the landscape (see Figure 17). Bobolink and Eastern Meadowlark occurrence was influenced by the underlying zoning, and the patches where these species are most likely to occur are aggregated in the areas zoned for agriculture or have rural-agriculture zoning. The patches with the highest probability of occurrence are also in areas where hayfields are more prevalent. For Eastern Kingbirds, Grasshopper Sparrows,
and Savannah Sparrows, which are more generalist species and are found in mixed farmland and can tolerate a wider range of vegetative structure, the probability of occurrence was less localized. Patches where these species were likely to occur were uniformly distributed throughout the study area. Horned Larks had higher probabilities of occurrence in the area where row crops are more dominant. The individual species models predict certain species are more likely to partition the landscape, but throughout the study area there is also likely to be overlap among the species, with multiple species using the same habitat patches.
Figure 17. Map of Predicted Occurrence for Individual Species. Probability of occurrence of individual focal grassland bird species within the study area for: a) Bobolink, b) Eastern Kingbird, c) Eastern Meadowlark, d) Grasshopper Sparrow, e) Horned Lark, and f) Savannah Sparrow.
**Landscape Diagnosis**

A landscape analysis of Chester County comparing grass-cropland land cover to residential land use revealed certain landscape patterns within the study area. Grass-cropland is the predominant land cover type in the study area followed by residential development. Grass-cropland covers 52 percent of the land area (32,148 hectares), and residential development covers 17 percent (10,363 hectares). The remaining area is comprised of woodlands and shrublands, urban development, roads, water, wetlands and open space (see Table 10). In the study area, there are 32,020 hectares of grass-cropland across 2,830 patches, which is the highest density of grass-cropland in the county. Comparing the mean area of grass-cropland patches ($\text{Area}_{\text{MN}} = 11 \text{ ha}$) to the weighted mean ($\text{Area}_{\text{AM}} = 93 \text{ ha}$), which places more importance on large patches, produces a coefficient of variation of about 267. The relatively large coefficient indicates distribution of patches based on size is comparatively high and patches are large on average, but there are also many small patches (Botequilha-Leitão and Ahern 2002). Among those patches almost half of the total grass-cropland land cover area is core area (the interior area of the patch 50 meters from the edge) that is divided among 2,650 discrete core areas with an average of 6 hectares per core area.
Patch connectivity can be measured according to a few different aggregation metrics. The clumpiness index (CLUMP) provides a measure of aggregation for the focal patch and is an effective measure of fragmentation (McGarigal 2015). A value closer to one indicates the land cover types are increasingly aggregated. Both grass-cropland (CLUMP<sub>G-C</sub> = 0.919) and residential land cover (CLUMP<sub>RES</sub> = 0.892) had values close to one. The patch cohesion index (COHESION) quantifies the connectivity of a landscape from the perspective of an organism moving within the landscape. Cohesion decreases with patch subdivision and disconnection and is related to the proportion of different land cover types bordering the focal patch (McGarigal 2015). As percentages increase, aggregation increases. Similar to the clumpiness index, these scores were high for both land cover types (COHESION<sub>G-C</sub> = 98.6 and COHESION<sub>RES</sub> = 95.5). Both of these measures suggest high levels of aggregation of land uses, which is evident when viewing
a density map of grass-cropland patches in the study area and throughout Chester County (see Figure 18).

![Map of Patch Density](image)

**Figure 18. Map of Patch Density.** Density of grass-cropland patches throughout Chester County and within the study area.

Nevertheless, when viewing these aggregation metrics, it is important to also assess them at the landscape scale to evaluate the spatial distribution of land cover types and the interspersion of different patch types—the spatial intermixing of different patch types that can lead to conflicting adjacencies. The contagion index (CONTAG) measures the extent of aggregation as a percentage of the maximum possible across the landscape, with higher values of contagion representing more like-cell adjacencies (i.e., having more
of the same land cover type cells touching) (McGarigal 2015). The contagion index for the study area is 61.6. This indicates there is a fair amount of interspersion, which means similar land cover types are aggregated but are not all in one area of the landscape.

Another factor that could impact the suitability of agricultural lands for grassland birds is increased human population density and the resulting development patterns. Using roads and developed parcels (i.e., residential, commercial, institutional, and industrial) as a proxy for human density, the landscape analysis shows about 24 percent of the study area is developed, with a development density more than two times greater than grass-cropland patch density. Much of the developed areas show a high degree of aggregation ($CLUMP_{\text{dev}} = 89.4$ and $COHESION_{\text{dev}} = 97.9$), with the greatest density of development surrounding the agricultural belt with little buffering area (see Figure 19).

With the county’s population expected to grow by 27 percent by 2045, an increase in population and development density in the study area could affect habitat occupancy if greater proximity to human disturbance (e.g., dogs, cats, and kids) depresses grassland bird populations. Figure 19a-d shows that though the focal species use the landscape differently and can be grouped according to their habitat use, almost all actual and predicted habitat occupancy does not overlap with the more densely developed parts of the study area.
Figure 19. Map of Development Density with Avian Data. Development density in relation to habitat occupancy models for the four representative species groups: a) Eastern Meadowlark group, b) Eastern Kingbird, c) Grasshopper Sparrow Group, and d) Horned Lark.
The Chester County Planning Commission has identified threats and opportunities related to land use change in the county (Chester County Planning Commission 2018c), which I analyzed in terms of their impact within the study area. The County identified undeveloped, unprotected parcels that are 10 acres (~4 hectares) or larger that are susceptible to land use change. The County also identified farmland preservation opportunities based on “existing land use, prime farmland soils, parcel size, and proximity to existing easements” (Chester County Planning Commission 2018c, 53). Within the study area, 10,584 hectares have been identified as threatened. Most of the susceptible parcels are in the growth areas (321 parcels, mean = 11 ha) and have not been prioritized for preservation though there are a few exceptions. There are 4,667 hectares that have been prioritize for preservation, almost all of which are within the rural resource area (4,544 ha). The prioritized land represents 9 percent of the threatened land (1,026 ha) and is divided among 60 parcels on 6,692 hectares (mean = 17 ha).

Applying these data to the grass-cropland patches within the study area shows the potential increase in aggregation of agricultural land use. Currently, 13,169 hectares are preserved (Area$_{MN}$ = 23 ha, S.D. = 39.7), and 2,761 hectares could be added to the existing patches, increasing the average proportion of preservation by almost 10 percent (mean = 0.078, S.D. = 0.187). However, conservation would be uneven among grasslands and croplands, with about 25 percent of the targeted lands being grasslands (719 ha). Though the average increase would be small (mean = 1.26 ha, S.D. 3.98), most of the new grassland preservation would be concentrated in less than 10 percent of the patches.

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19 This analysis relates only to those parcels that fall within the 750 sq. km study area though the CCPC’s land use analysis encompasses the entire county.
and would fill in preservation gaps on patches that have large contiguous areas of grassland or would buffer croplands.

There are more patches susceptible to land use change in the southwestern part of the study area. This is also an area that does not have protective agricultural zoning. In this area, there is a higher density of threatened patches, but there is also a higher concentration of patches that have been targeted for preservation.

Overall, applying the avian grassland linear model, shows the parcels targeted for preservation have a high level of overlap with habitat that is predicted to have at least one focal species. This overlap is an indication of efficiency of preservation efforts with the goal of developing multifunctional landscapes (Bishop and Myers 2005) (see Figure 20). However, benefits associated with land preservation in the study area would be unevenly distributed among the focal species. Focal species were grouped according to their habitat associations to determine how preservation projections would impact individual species (see Figure 21).

Based on model probabilities, Horned Lark habitat has the greatest overlap with parcels that have been prioritized for preservation. For habitat patches that had a 60 percent or greater probability of occupancy by Horned Larks, 1,740 hectares are threatened, but 1,379 hectares have been targeted for preservation. The outlook is quite different for Eastern Meadowlarks and Bobolinks. For these species, high probability patches are concentrated in a single area that already has a high level of preservation, and there are few patches that are either prioritized for future preservation or that are threatened. The threatened area with these high probability patches is only 13 hectares, and total additional preserved area would be 156 hectares.
Grasshopper Sparrows and Savannah Sparrows would have similar outcomes though they would be expected to be more widely distribution. Within that predicted range, there are more preservation opportunities than threatened patches (213 ha and 81 ha, respectively). Eastern Kingbird, which is found in diverse grassland habitats, has a high probability of occurring in patches across the study area. As such, it is likely to be in areas that already have a high proportion of preserved lands and those that offer more opportunity for preservation. Predicted patch occupancy for Eastern Kingbird patches overlaps with 904 hectares in prioritized parcels and 708 hectares in threatened parcels that are concentrated in a few areas. The avian data only provides probabilities of occupancy, but in areas that are not preserved, they represent potential loss of habitat, which could affect occupancy levels. Preservation offers more stability in habitat across the landscape because the threat of development is removed.

In the next chapter I compare my findings to other grassland bird studies and discuss the limitations of my models. Additionally, I evaluate the implications of my models on future land preservation and conservation efforts in Chester County. Finally, I discuss management and policy changes that could improve grassland bird conservation efforts.
Figure 20. Map of future land preservation in Chester County. The map shows the different land cover types that will be preserved and areas that have been permanently protected (data source: Chester County Planning Commission 2018a, 2018b).
Figure 21. Map of Predicted Habitat Use with Projected Preservation. Predicted use of available habitat patches in a landscape depicting projected preservation of grasslands and croplands for: a) Horned Lark, b) Eastern Meadowlark and Bobolink, c) Grasshopper Sparrow and Savannah Sparrow, and d) Eastern Kingbird. The inset box shows patches with a high probability of occupancy (> 0.60) and parcels where preservation has been prioritized and parcels likely to undergo development.
Influence of Landscape Factors on Grassland Bird Occurrence

Diversity Model

The field surveys and the prediction models provide insight into grassland bird habitat usage in southern Chester County’s agricultural and rural landscapes. Though grassland birds in Chester County are not as abundant as generalist bird species, census data showed they were detected more frequently than during the second Breeding Bird Atlas (Wilson et al. 2012). Higher detection rates are most likely related to the sampling method. Grassland birds have been shown to avoid habitat edges (Bock et al. 1999; Bollinger and Gavin 2004; Renfrew and Ribic 2008; Sliwinski and Koper 2012), and with BBA surveys conducted primarily through road-based samples supplemented by point counts within region blocks, relative abundance and frequency of occurrence are likely to have been reduced, which would affect detection probabilities (Wellicome et al. 2014). With all sampling points located more than 50 meters from the edge of the patch during the 2014-15 survey, it would be expected that count statistics would be higher.

The total species model suggests grassland bird habitat occupancy is influenced by local and landscape variables. The results support the findings of other studies that have evaluated the influence of area on habitat occupancy (Vickery et al. 1994; Herkert 1994; Johnson and Igl 2001; Bakker et al. 2002) though this study measured occupancy rates in terms of core area rather than total area for five of the focal species. With the exception of Eastern Kingbirds, the other focal species are edge-sensitive, so total area is likely to be less important than the amount of interior area available. Use of the core area
metric is supported by previous theoretical research (Ewers and Didham 2007) and Herse et al.’s (2017) study of grassland birds that found availability of core habitat may be a more important factor for edge-sensitive species than total area. The number of focal species in a patch was positively associated with the amount of core area. Projected occupancy rates were higher in larger patches and when a higher percentage of the patch was farther from the edge. Both measures of vegetation, which categorized patches according to vegetation structure, were also significant factors in the models and provided insight into how species within the focal group partition the landscape.

Patch context within the landscape is another important factor in habitat occupancy. Similar to the Hamer et al. (2006) study, my model showed aggregation of patches, measured by the distribution and size of habitat patches, had a positive effect on the number of focal species predicted to occupy a patch. Having larger habitat patches clustered together across the landscape limits isolation of patches, which can disrupt movement among local populations (McGarigal 2015). The degree of fragmentation has implications for metapopulations across the landscape—with less isolation, populations can move among patches, stabilizing the population between source and sink areas; however, there is a habitat loss threshold at which the landscape becomes too disconnected to support populations (With et al. 2006).

Within the landscape, persistence of bird populations may depend on having enough connectivity between source and sink habitats, which can be dependent on habitat quality (i.e. fragmentation which increase edges and disturbance such as agricultural intensification) as well as availability. Scheiman et al. (2007) found Bobolinks in west-central Indiana had a dispersal rate sufficient to maintain a patchy metapopulation and
concluded that having adequate connectivity can affect regional persistence of species. Murphy (2001) studied an Eastern Kingbird population in central New York that inhabited a mosaic of habitats and determined lower quality sinks occupied by the birds stabilized population in high conservation value patches. By monitoring the territory in the sources, birds were about to move into openings as they became available. Proximity, being a significant predictor variable suggests the focal species occurrence, is influenced by the availability of surrounding patches. This indicates metapopulation dynamics may influence single patch occupancy within this study; therefore, occupancy of high value patches may support populations on lower quality patches.

*Species Models*

Occupancy rates of individual species varied throughout the landscape. Bobolinks and Eastern Meadowlarks clustered around the area where hayfields comprise a higher proportion of the landscape. Horned Larks occurred most frequently in the region dominated by croplands. Eastern Kingbirds, Grasshopper Sparrows, and Savannah Sparrows were distributed more evenly through the study area. These habitat occupancy patterns reflect vegetation preferences, which were important predictors for all species except Eastern Kingbird and Savannah Sparrow. These findings were consistent with other studies that have shown the influence of vegetation structure on grassland birds habitat occupancy and the importance of varied vegetative characteristics, such as height and density, for different species (Vickery et al. 1994; Herkert 1994; Best et al. 2001; Bakker et al. 2002).
For all focal species except for Eastern Meadowlark, core area or a parameter that quantifies the percentage of the patch that is comprised of core area, were significant predictors of habitat occupancy. Grassland birds have been found to be area sensitive, and these species models generally support the findings of other researchers. The Bobolink regression model supports earlier studies (Vickery et al. 1994; Herkert 1994; Bollinger 1995; Johnson and Igl 2001) though the measurement in this model was based on the proportion of the patch that is core area rather than a measure of the patch’s total area.

Bakker et al. (2002) found Bobolink occurrence in eastern South Dakota was related only to vegetation variables, but Bollinger (1995) found Bobolink occurrence in central New York was associated with a combination of measures of area and vegetation. The Bobolink regression model for this study supports Bollinger’s model, but it is similar to Winter et al. (2006), which found landscape factors were also significant predictors. In this model, proximity to other similar patches, which is a measure of patch isolation, was a significant predictor of occupancy. But when patch level vegetation data were removed from the model, the interaction between proximity and the level of preservation became significant. Since high levels of preservation provide aggregation benefits by increasing connectivity (Rissman and Merenlender 2008), the relationship between preservation and proximity likely reflected the benefits of aggregation as they relate to habitat suitability. With high levels of both variables, development is limited and there is less fragmentation. When applying the model to the landscape, the underlying zoning was also an important predictor of Bobolink presence. Other studies have not looked at the influence of zoning
on grassland bird species, but these results suggest more restrictive zoning that limits land use and the density of development may help to reduce habitat isolation.

Unlike Bobolink, findings for Eastern Meadowlark occurrence have been mixed. Herkert (1994) and Vickery et al. (1994) found Eastern Meadowlark to be area sensitive. Bollinger (1995) and Winter and Faaborg (1999) and Johnson and Igl (2001) and Bakker et al. (2002), who studied its congener, the Western Meadowlark (*Sturnella neglecta*), determined area sensitivity did not influence patch occupancy. The results of my model support the latter findings. Eastern Meadowlark occupancy was driven by patch level vegetation data, which supports the findings of Vickery et al. (1994) and Bollinger (1995). Eastern Meadowlarks also benefited from habitat connectivity. The other predictive factors for occurrence were proximity and levels of preservation though these variables did not have an interactive effect as they had with Bobolinks. When patch level vegetation was removed from of the model, the underlying zoning became a predictive factor, which, again, points to the importance of patch aggregation and limiting external influences from the landscape matrix. These findings are consistent with Bock et al. (1999) who determined several of the focal species (i.e., Bobolink, Grasshopper Sparrow, Savannah Sparrow, and the congener species, Western Meadowlark) avoided suburban edges.

Vegetation measurements and the percentage of core area were also predictors for Horned Lark though this species has different habitat requirements. Horned Lark were negatively associated with vegetation measurements that increase with the amount of grass in the patch. There was, however, conflicting evidence regarding area sensitivity based on the positive association with higher percentages of core area and the positive
association with shape complexity and the interaction between shape and vegetation. In contrast to Davis (2004), who found Horned Lark abundance in Southern Saskatchewan increased as edge to area ratio decreased, greater shape complexity, and subsequent exposure to edge, was a positive predictor of occupancy in this study’s model. When shape interacted with vegetation scores, increased complexity and low vegetation scores had higher probability outcomes.

Because habitat patches were identified using the Anderson Level IV Classification System (Anderson et al. 1976) without regard to landownership and parcel boundaries, the majority of habitat patches encompassed multiple parcels. Horned Larks have been shown to have an affinity for row crop fields (Best et al. 2001). In the region predominantly comprised of cropland, parcel density is higher, and with increased patch development associated with the addition of each new parcel, patch complexity increases because there is more habitat edge and perforations associated with development of the property (Botequilha-Leitão and Ahern 2002). The model suggests Horned Larks may not be seeking edge; rather, their occupancy may be driven by vegetation preferences.

When patch level vegetation was removed from the model, the underlying zoning was a significant predictor. There was a negative relationship between Horned Lark occurrence and the more restrictive agriculture and rural-agriculture zoning districts. Once again, this relationship likely reflects the location of the majority of croplands in municipalities that do not separate agricultural uses from other land uses.

Few grassland bird studies that include Eastern Kingbirds have found conclusive habitat predictors. Johnson and Igl (2001), whose results were based on few detections of Eastern Kingbirds in the northern Great Plains, found area had a positive effect on
occurrence. Murphy (2003) found Eastern Kingbird population trends for eastern and central U.S. were correlated with the amount of available habitat across the landscape, specifically the presence of hayfields. Though the model for this study only accounted for 15 percent of the variance, total patch area was a positive predictor of species presence. Additionally, the percentage of core area was important when the surrounding matrix increased patch isolation and edge contrast. When patches were more clustered, core area became less important. Eastern Kingbirds are edge species that nest in trees within agricultural fields (Murphy and Pyle 2018), so they are likely to be less sensitive to edge effects that reduce the occurrence of other grassland bird species.

The relationship between Grasshopper Sparrow habitat occupancy and area has been well-established (Vickery et al. 1994; Herkert 1994; Bollinger 1995; Bakker et al. 2002; Davis 2004) though Johnson and IgI (2001) noted regional variability in area sensitivity, and Winter and Faaborg (1999) found density was dependent on vegetation rather than area. The models from this study support findings of area sensitivity, with prediction rates of Grasshopper Sparrow occurrence increasing with the percentage of core area within a habitat patch. Similar to Vickery et al. (1994) and Davis (2004), this study showed patch level vegetation influenced predicted occupancy rates—patches with a greater proportion of grasses were more likely to be used by Grasshopper Sparrows. When vegetation was removed from the model, proximity became a significant predictor, indicating a sensitivity to patch isolation. The positive association with proximity relates to fragmentation and is aligned with the research of Herse et al. (2017), which found Grasshopper Sparrows in eastern Kansas favored landscapes comprised of large grassland areas or unfragmented grassland with fewer edges.
Similar to Grasshopper Sparrows, Savannah Sparrows have consistently been identified as area sensitive (Vickery et al. 1994; Herkert 1994; Bollinger 1995; Bakker et al. 2002) though Davis (2004) found they were area insensitive but edge avoidant. In the model from this study, core area had the greatest influence on patch occupancy. Fragmentation of the core area was also a factor as measured by the number of distinct core areas. As core area increased, if the core area was separated into discrete areas of the patch, probability of Savannah Sparrow occurrence decreased. This may be the case because an increasing number of core areas could break up the continuity of the total core area effectively cutting a large patch into smaller patches.

**Model Assessment**

There were limitations to the models developed during this study. The first set of limitations relate to the field sampling protocol and data collection. Though point counts are widely used (Buckland 2006) and have been used in similar studies (Vickery et al. 1994; Johnson and Igl 2001; Best et al. 2001; Davis 2004; Lockhart and Koper 2018), other researchers have used strip transects for sampling in which the observer moves along a transect and records all birds to a fixed distance (Herkert 1994; Bollinger 1995; Bakker et al. 2002; Winter et al. 2006). Researchers have found both types of surveys produce similar results when measuring species richness (Manuwal and Carey 1991; Verner and Ritter 1985; Taulman 2013) though point counts provide some advantages
related to efficiency and control of duration (Verner and Ritter 1985) and mean number of birds detected (Taulman 2013).20

A potentially confounding issue with point counts is detection inefficiency related to distance and density that leads to “false absences” (Dunham and Rieman 1999). If birds did not call, sing, or show themselves within the 10-minute censusing period, they would not be recorded. Also, detectability can be reduced by more than 50 percent when a bird occurs farther than 50 meters from the observer (Diefenbach et al. 2003). As a result, species may go undetected leading to an underestimation of species presence. Undercounting can also be species-dependent. Sound travels at different distances, depending on vegetation and atmospheric conditions (Yip et al. 2017), and some bird songs are more difficult to hear based on the frequency and attenuation of the song (Simons et al. 2007). For example, Grasshopper Sparrows, which have a very high frequency song (mean frequency = 8,600 vibrations per second) (Bent 1968), could have a lower detection rate than Eastern Meadowlarks, which have a lower frequency song (mean frequency = 4,400 vibrations per second) that carries over a distance (Bent 1958). Lower detection rates can reduce the predictive power of the models.

There were also limitations related to data collection. To maximize survey coverage within the study area, I conducted one survey per census plot. This enabled me to gather more wide-ranging data—surveys were conducted on 37 percent of the available habitat patches—but I was unable to collect multiple sets of data for each site.

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20 Both methods produce reliable results when recording presence or absence of a species; however, researchers have raised concerns about the reliability of these methods for estimating avian density and abundance (Rosenstock et al. 2002; Diefenbach et al. 2003), which can reduce statistical power when attempting to document population trends over time.
which would have provided insight into temporal factors related to site suitability and could have provided more opportunities for detection. To validate my rate of detection, I double-sampled a subsample of patches during both field seasons, returning to about 10 percent of the patches in 2014 and 2015. Within the two-year validation set, observations were consistent, with an average of 87 percent of the observations matching between the initial field visit and the validation visit. The validation visits demonstrated the accuracy of the surveys but did not strengthen the models’ ability to predict trends.

Additionally, I generally conducted the surveys during the safe dates in which a species is likely to be breeding. The majority of safe dates for the focal species overlap (Wilson et al. 2012); however, some observations of focal species were made outside of the safe dates. This could account for lower detection rates of some species, and some detections may have counted birds that were not breeding because they were migrants or, later in the season, they may have been non-breeding juveniles. Nevertheless, no observations were discarded since the focus of this study was not solely on breeding success because a patch supporting a focal species, either through nesting or foraging, is part of the grassland habitat network and contributes to overall habitat suitability within a metapopulation model.

Another issue that could influence the models is survey effort. Data collection took place over two years, which matches or exceeds the survey effort of similar grassland bird studies (Vickery et al. 1994; Johnson and Igl 2001; Best et al. 2001; Bakker et al. 2002; Davis 2004; Lockhart and Koper 2018). Although other grassland bird studies that have been conducted over three years (Herkert 1994) or four years (Winter et al. 2006) have found responses to habitat that were not consistent over the
study period, having only two years of observations limits the ability to identify grassland bird habitat occupancy trends over time and could lower detection rates if events or environmental conditions of a single year affect occurrence in that year.

Uneven detection rates among the focal species may indicate differences in abundance and distribution, but without more years of observation, it is not possible to make any definitive determinations. Additionally, because census data were based on the presence or absence of the focal species, it is only possible to make rough approximations of habitat suitability by showing habitat associations. Long-term demographic data would help establish a more precise understanding of factors related to maintaining viable populations (Herkert 1994).

The $R^2$ values of the models ranged from 0.154 to 0.603, with several being relatively low. These low $R^2$ values reflect the nature of land models and their attempt to impose uniform metrics on the landscape. Though the landscape is comprised of individual patches that are relatively homogenous, there is a degree of internal heterogeneity (Forman 1995). Across the landscape and within patches, habitat quality can vary according to vegetation, microclimate, soils, water access, and the presence of predators. Because habitat parameters are numerous and vary spatiotemporally, it is difficult to measure all significant variables.

A lower $R^2$ value indicates there are contributing factors that the models did not capture. Thus, habitat may not be the only controlling factor for occurrence. For example, these models did not include human population density data. There may be an upper limit to human population density that precludes grassland bird occurrence regardless of habitat availability. I used zoning as a proxy for development, but including density by
township or some other measure of the existing development pattern could have provided more insight into how human settlement patterns affect habitat occupancy. Given the wide range of potential factors, these models can only account for a portion of the variance.

Although the lower $R^2$ values for the models generated for this study limit the utility of the models for predicting habitat occupancy in other regions, they can help to identify important variables for habitat occupancy. The utility of these models is they demonstrate the relationships between the focal species and the grass-cropland landscape they occupy. It should also be noted, the $R^2$ values were comparable to or larger than the values generated from similar studies (Vickery et al. 1994; Bakker et al. 2002; Murphy 2003).

Despite the models’ shortcomings, the results support previous research findings. Peterjohn (2003) noted that confirming earlier studies is important because when similar positive associations between landscape variables and populations are found at a range of geographic scales, those associations may suggest “important clues” toward positively identifying influences on population changes. Additionally, the bird census complemented earlier BBA censusing by expanding the number of census plots within the BBA survey area in southern Chester County, enhancing grassland bird population estimates in the county and providing more insight into habitat usage. The models also provide new insights related to the development of a grassland reserve network. The models incorporated preservation and land use variables in addition to biological and landscape data, which provided an opportunity to analyze the effects of zoning and previous preservation efforts on the focal species that use the grass-cropland network in
Chester County. Incorporating land use data allowed for the evaluation of different zoning regulations and levels of preservation and how they relate to habitat isolation and the implications for grassland avian assemblages.

These models may have limited utility beyond the Chester County landscape because certain factors are likely to be specific to the study area (e.g. patch complexity and less restrictive zoning increase the probability of Horned Lark occurrence because they are indicative of a patch’s location within an area dominated by croplands). But they demonstrate there is a relationship between land use policy and ecological function. Though much effort has been put into preservation of the agricultural landscape in the study area, no research has been published on the impacts of these efforts. My research findings go beyond patch-level conservation and could be useful in extending the current “toolbox” for strategic preservation planning.

Implications for Planning

The greatest density of grassland breeding bird populations is in the Midwest and the Great Plains (NRCS 1999); thus, much of the research and conservation initiatives have focused on those regions. However, there is a case to be made for prioritizing the conservation of grassland birds in the Northeast. One major reason is to conserve biodiversity. Though densities of grassland birds are lower in the Northeast, the region has relatively high densities of eastern subspecies of Savannah sparrow (P. s. savanna), Grasshopper Sparrow (A. s. pratensis), Horned Lark (E. a. praticola), and Eastern Meadowlark (S. m. magna). For Bobolink, which are more evenly distributed, the
Northeast has more than 10 percent of the total breeding population (Wells and Rosenberg 1999).

Based on the model outcomes from this study and the results of similar studies, it is feasible that the Northeast could also provide habitat for low density populations that maintain these avian assemblages throughout their native and expanded ranges (Askins 1999, 2002). Maintaining these birds in the Northeast may provide important use of alternative habitat that will enable long-term persistence. Ward et al. (2018) looked at changes in bird populations over a 100 year period and found that alternative habitats, those of medium and low affinity, can provide opportunities for expansion when high quality habitat is already occupied. This expansion can lead to an increase in population size. They argued long-term persistence may depend on the availability of alternative habitat. Finding opportunities for habitat expansion is increasingly important as grassland bird populations continue to decline.

In the Northeast, there is no single, physiographic region with dense grassland bird assemblages, so it would be more effective to create regional plans that develop initiatives around specific species that can be supported in a particular landscape (Wells and Rosenberg 1999). Establishing a regional conservation plan would require cooperation among public and private entities because long-term persistence may not be possible if grassland bird conservation is limited to a few isolated sites on public land (Wells and Rosenberg 1999). Enlisting private agricultural lands would help to ensure the availability of aggregated habitat. Though there are drawbacks to relying on agricultural lands, they can provide alternative habitat to grassland birds that do not have access to high quality habitats and can provide regional connectivity with higher quality habitat. In
the Northeast, grassland birds have adapted to highly artificial habitats and can be maintained if working landscapes are managed to include space for birds (Askins 2002).

A functional landscape for grassland bird conservation would be a mosaic of agricultural lands and grasslands that create a buffering landscape that limits fragmentation (Best et al. 2001; Lockhart and Koper 2018). Intermixing grass-like land cover types, such as pasture, alfalfa hay, and filter strips, with other agricultural land cover can make an agricultural district more habitable for bird species that prefer more cover (Best et al. 2001). Row crops can be a part of the mosaic as well. Croplands would seem to offer poor quality habitat for most grassland birds, but birds have been shown to use row crops either for nesting or foraging (Best 1986; Best et al. 1995; McLachlan et al. 2007; VanBeek et al. 2014). Given the different habitat affinities among grassland birds (Ward et al. 2018), having diversity among agricultural land uses would be an important factor for encouraging species richness. There can also be a diversity of patch sizes above a minimum threshold because focusing too heavily on large patches could preclude the conservation of smaller patches that have value (Ribic et al. 2009). Area sensitivity is a guiding factor, but connectivity and proximity have also been shown to influence habitat occupancy. Therefore, reduction in patch size may not be as influential as overall fragmentation.

Strategic planning will play a major role in developing and maintaining the needed grassland and working landscape network to sustain grassland bird populations. The most effective planning would include a regional plan that integrates bird conservation with existing land preservation and land use regulations. Implementing agricultural zoning will be necessary, but it will be less effective on its own because
agricultural zoning can be “un-done”—zoning can be replaced by weaker zoning, and rezoning of individual parcels can weaken agricultural zoning over time (Kruft 2001). Including land preservation in the comprehensive planning process can supplement strong zoning by providing permanence in the landscape (Daniels and Lapping 2005). Landscape permanence removes the specter of land use change known a “impermanence syndrome,” which occurs when population pressure increases the amount of development in farming areas, and farmers’ decisions are guided, in part, by the possibility of eventually selling the farm for large profits (Coughlin and Keene 1981). This can create instability because there is less certainty that the local agricultural industry will persist. Ideally, land trusts and local governments would focus on preserving land strategically, such that efforts are guided by a broader framework that considers individual properties within the context of the greater network.

Science-based prioritization tools are available for broad-scale planning (e.g., The Nature Conservancy’s Conservation by Design, Partners in Flight’s North American Landbird Conservation Plan), and landscape-scale plans, such as Audubon’s Important Bird Areas and the Pennsylvania Natural Heritage Program (PNHP), can help guide preservation planning. For example, the PNHP provides scientific information and assistance to support the conservation of biological diversity. PNHP data can be used to guide conservation efforts, land-use planning, and land development (PA DCNR 2019). Using these planning tools enables planners to link preservation decisions to create ecoregions. These tools can also assist with assessing the conservation values of opportunistic projects that arise when landowners propose a conservation plan (Wilson 2011). Taking a landscape-scale view could help smaller land preservation organizations
and local governments prioritize preservation efforts by highlighting zones of richness (Bishop and Myers 2005) while simultaneously identifying areas where expensive preservation opportunities should be forgone because they are better suited for development (Underwood et al. 2011).

Securing funding for these projects is crucial. Funding from state natural resources agencies, such as grants for land preservation through Pennsylvania’s Department of Conservation and Natural Resources, can be made to local governments or land trusts. The Pennsylvania Agricultural Conservation Easement Purchase Program provides funding to county governments to purchase agricultural conservation easements on farmland. In Chester County, as of 2017, this program has preserved 327 farms on 27,800 acres (11,250 ha) (Pennsylvania Department of Agriculture 2018).

Though large landscape-scale guidance will be helpful, much of the preservation will be executed by regional and local land trusts and local governments that will have finer scale priorities. They will need a practical model to integrate broad-scale objectives with specific criteria for relatively small-scale conservation. Using a framework like the one I have developed can help to ensure planning is strategic rather than opportunistic. The model can be used to evaluate how well parcels align with the objectives of the strategic plan and help in setting priorities for preservation. Once high-quality, ecologically connected habitat has been identified and prioritized, local planners can use this research to develop goals related to preservation and land management. (Appendix A provides an example of a planning tool that could be used to evaluate the degree to which an organization is engaging in strategic planning.)
Preservation, however, cannot address all conservation land use objectives. Land preservation would work best in conjunction with comprehensive planning. Ideally, the comprehensive plan would describe the complementary nature of the persistence of working landscapes and grassland bird conservation in the Natural Resources Inventory, Economic Base, and Land Use sections with specific recommendations for strategies to support both (Daniels 2014). Zoning could then be developed in conjunction with the comprehensive plan to reinforce these goals.

Zoning can be used to address growth and conservation. First, zoning can promote compact growth away from working landscapes and the habitats they support because planning should not focus on protection without identifying the best areas to develop (Underwood et al. 2011). Then, to prioritize agricultural land uses, strong agricultural zoning could be implemented. Daniels (1997) recommends zoning of at least 1 building lot per 20 - 25 acres (8 – 10 ha). The zoning can be designed with a fixed area ratio that allows one dwelling unit for a specified area (e.g. 1 dwelling per 25 acres on a lot of 2 acres or less) or on a sliding scale that allows the construction of non-farm dwellings based on the size of the existing parcel with fewer dwellings allowed as the size of the parcel increases (Daniels 2014). In addition, agricultural zoning can be non-exclusive to allow other uses, which can make it easier to implement (Hartzell 1999) though it can also weaken protections.

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21 Using a sliding scale keeps a larger contiguous, unbroken parcel of land in agricultural use while maintaining developmental opportunities on smaller parcels, which can enable municipalities to accommodate growth without fragmenting agricultural lands (Hartzell 1999); however, it can also be seen as favoring rural landowners over farms with large farms (Daniels 2014).
In municipalities that do not have agricultural zoning, it can be used to add a layer of protection within the zoning district. Agricultural zoning can protect large tracts of land throughout a municipality at a relatively low cost, and it is an alternative to focusing on parcel by parcel preservation (Kruft 2001). This zoning can also protect high-quality soils, buffer agricultural uses from conflicting land uses, reduce the conversion of farmland to other uses, and limit fragmentation of working landscapes and habitat (Daniels 2014). In municipalities that already have relatively weak agricultural zoning (i.e., 1 to 10 acre minimum lot size), the zoning can be strengthened to require larger minimum lot sizes.22

Rural zoning, which is in place in much of the study area in Chester County, is not a good alternative to agricultural zoning when trying to preserve agricultural land. Though rural zoning may require large lots, it is not as effective because the emphasis in on protecting the “rural character” of an area and each development proposal is considered individually, rather than as part of a comprehensive plan (Daniels 1997). Therefore, rural zoning can result in clusters of suburban communities with some open space between them instead of a viable working landscape. However, if it is politically unfeasible to implement stronger zoning, farmers can create Agricultural Security Areas (ASAs) (Hartzell 1999). ASAs enable farmers to voluntarily establish a layer of protection from development. With ASAs, a farmer or group of farmers, who own 250 acres (100 ha) or more of agricultural land and have a specified income from the land (e.g., $2000 in Pennsylvania), can petition the local government to include the land in an

22 Agricultural zoning has been upheld at one dwelling per 50 acres, see Codorus Township v. Rogers, 492 A. 2d 73 (Pa. Commw. 1985).
ASA for a seven year period (Pennsylvania Department of Agriculture 2019). ASAs can reduce the likelihood of farmers having restrictions placed on their farming practices or of having public nuisance conflicts arise.

In addition to zoning, other planning tools can limit disruption within working landscapes. Subdivision regulations can require minimum setbacks for development to buffer agricultural lands and separate development from the agricultural uses. Subdivision regulations can also require a certain percent of a tract slated for development be kept in open space, which is another way to provide a buffer between residential and agricultural uses (Daniels 2014). These types of subdivision regulation are most useful when residential and agricultural land uses are adjoined. The capital improvements program (CIP) can be an effective tool for ensuring development conforms to the conservation objectives of the comprehensive plan, with infrastructure spending that does not weaken conservation planning by making it easier to build in less developed areas (Brody and Highfield 2005). Incorporating conservation goals during all the phases of general land use planning is critical to minimizing conflict between future growth and conservation (Underwood et al. 2011).

Strong land use regulation can not only protect agricultural activities and the interests of farmers (Daniels 2014), it can also help to shape landscape spatial patterns and create a more open landscape, which is associated with increased habitat occupancy by grassland birds (Davis 2004; Ribic et al. 2009). These measures could be important for grassland birds that exhibit area sensitivity and tend to avoid smaller patches because strategic preservation and zoning will help to aggregate the working landscape, creating a landscape that buffers smaller patches, so they are more suitable to grassland birds.
However, this is often not politically possible. Fragmentation of local
governments can lead to disjointed land use decisions for adjacent areas, which can result
in planning that is incompatible and undermines long-term planning efforts. Adhering to
regional planning guidelines can reduce this threat, but regional planning is voluntary
because regional planning authorities do not have legal power to enforce plans.
Nevertheless, to be most effective, municipalities need to coordinate their planning
efforts (Daniels 2014).

Another challenge is lack of political will to use land use regulations to preserve a
rural economic base. Farmers may want the option to sell their land (Daniels 1997), and
municipalities may proceed cautiously because land use regulations have resulted in legal
battles and have not always been popular politically (Daniels and Lapping 2005). Though
there are challenges, it will be necessary to overcome them to support a viable
assemblage of grassland birds because agricultural protection is linked to habitat
conservation.

**Beyond Preservation**

Aggregation of the landscape through preservation and zoning will inevitably
have its limitations. Agricultural intensification, characterized by higher levels of
fertilization, less grazing, and a more frequent mowing regime (Allan et al. 2014), has
made agricultural lands less suitable for grassland birds (VanBeek et al. 2014). There are
a number of negative outcomes associated with agricultural intensification, but modifying
the timing of certain practices (e.g., mowing, fertilization, animal stocking densities) can
provide for biodiversity (Allan et al. 2014) and multifunctional landscapes (Frei et al.
Multifunctionality is the pursuit of multiple outcomes from a discrete land area with productivity measured by commodity and non-commodity outputs, such as ecosystem services and increased biodiversity (Ahern 2013). Avian biodiversity is a significant predictor of multifunctionality and could be used as an indicator of how well agricultural fields are meeting multiple criteria (Frei et al. 2018).

Because mowing regimes can be particularly detrimental to breeding grassland birds, nesting success depends on minimizing disturbance during the breeding season. Hay cutting dates have shifted over the last 60 years and are now taking place earlier to coincide with peak nesting season (Herkert 1997; Nocera et al. 2005). Earlier mowing causes high rates of nest failure when nests are destroyed by haying machinery or by increasing exposure to predators (Bollinger et al. 1990; Perlut et al. 2006). Research has been done on the impacts of haying on nesting cycles to explore options for reducing the threat to reproductive success.

Studies done in the Champlain Valley on the New York-Vermont border have looked at the effects of early-, mid-, and late-cutting schedules, mowing before a cut-off date, and leaving areas uncut. Perlut et al. (2006) found Savannah Sparrow and Bobolinks had greater nesting success when fields were cut mid- to late-June, fledging about 2.5 more young than in early-cut fields (May 27 to June 11). Fields cut later offered an opportunity for birds to gain territories and complete nesting if nesting had been unsuccessful in fields that had been cut earlier. Based on those results, Perlut et al. (2011) developed a model in which farmers received an incentive through EQIP to mow before mid-May followed by a 65 day moratorium. Early haying produced a high-quality harvest in late May and allowed birds to rebuild nests if their nests had been disturbed. They
were able to go on to complete nesting activities and fledge young from fields that had been sinks. Under this program, songbird nesting success improved considerably. Masse et al. (2008) examined the relationship between leaving uncut areas during haying and predation. They found uncut areas offered greater protection for nests and reduced the risks associated with nest predation. There was also evidence that these buffering strips provided “resilient” patches where birds could rebuild nests if they have been destroyed.

The findings on manipulating haying schedules are encouraging and could have positive effects on grassland bird populations, especially in the Northeast.

One barrier to changing haying times is education. The relatively high levels of nesting success in these studies contradict the conventional thinking on conserving grassland birds in artificial hayfields in the Northeast (Perlut et al. 2006). The case has been made that fields cut during the breeding season are sinks, and hay should not be harvested until mid-July to early August to ensure nesting success (Atwood et al. 2017; NYSDEC Bureau of Wildlife 2019). However, these windows are overly restrictive and, most likely, unnecessary. Conservation organizations have begun to incorporate new information on haying during breeding season (Ochterski 2006; USDA, Natural Resources Conservation Service 2010), which may help to reduce tensions around grassland bird conservation.

Another barrier relates to the financial burden of conservation and ensuring farmers are not experiencing economic loss. Timing of hay cutting is driven by the need to balance yield and quality, which is measured by crude protein and acid detergent fiber (used to measure digestibility). There is evidence that delaying hay-cutting should not reduce the value of the hay. Nocera et al. (2005) found the nutritional quality of hay does
not decline below feeding requirements for beef-cattle if haying is delayed during peak nesting season, but postponing cutting by 1.5 weeks increases fledging rates. In addition, there is an economic value to cutting hay later because the seed heads are able to mature and reseed, creating a healthier hay stand with fewer weeds (J. Hicks, pers. comm.). Early haying also produces high protein hay (Perlut et al. 2011).

Integrating bird conservation with agricultural practices will have benefits, but achieving conservation outcomes will require cooperation among the scientific, policy, farming, and conservation communities. “Bird friendly” management requires an understanding of the logistical and financial costs associated with different options for management and financial support to offset additional costs associated with changing regimes (Burger 2006). To ensure resources are used efficiently, it is necessary to have a clear idea of the costs and benefits associated with changes in management and the most efficient funding sources, which can come from the government or alternative sources (Ciuzio et al. 2013). Perlut et al. (2011) found farmers were interested in agri-environmental programs that met their economic needs and were convenient to implement. Private landowners and public land managers need realistic guidelines for management practices that are beneficial to grassland birds and incentives to implement them. With funding available through the Farm Bill, there is an opportunity to implement management regimes that are beneficial to grassland birds. The next step is determining how to maximize positive benefits from these subsidies. Nevertheless, expectations must be realistic; if term-limited incentives are being used, such as those allocated through the Farm Bill’s Conservation Reserve Program (CRP), it is necessary to consider losses in
conservation value if reenrollment does not occur (Dayer et al. 2018).\textsuperscript{23} It will also be helpful to remember that the majority of conservation benefits can be achieved at a relatively low cost—it is the insistence on “getting it all” by maximizing the economic or ecological benefits that becomes extremely expensive (Polasky et al. 2008).

Peterjohn (2003, 17) concluded, “effective conservation of farmland birds will require innovative solutions based on current agricultural practices that benefit the greatest diversity of farmland birds.” Given the challenges to conserving grassland birds and the research on grassland bird preferences and behavior (Best et al. 2001; Scheiman et al. 2007; Ciuzio et al. 2013), we must find least-cost plans with the highest ecological returns. Using CRP lands to conserve grassland birds has been one major initiative of the Farm Bill, but the implementation of this program may be flawed. Taking small fields out of production via CRP may have limited value for edge sensitive species. A more effective strategy might be to “leave some for the birds,” such that more resources are put into planting linear strips instead of blocks unless the blocks are substantially greater than 10 hectares. In croplands, planting linear filter strips, which have a range of soil and water quality benefits, can be used to supplement grass-like habitat among row crops. Beyond the Farm Bill resources, another strategy might be to engage in early haying or delayed haying and using preservation efforts to focus on “infill” buffering of patches that are already protected.

\textsuperscript{23} “Persistence,” is the term used to describe whether landowners continue conservation behaviors after term-limited financial incentive payments end. Persistence is often assumed, but long-term goals can be diminished if it is not achieved.
Planning for Active Grassland Bird Management in Chester County

Effective grassland bird conservation requires the protection and enhancement of artificial grassland habitats (Vickery et al. 1999). In Chester County, preservation efforts have yielded a large contiguous area of protected lands. On the whole, landscape metrics indicate the grass-cropland network within the study area could provide suitable habitat for grassland birds. The study area, which is also the most concentrated agricultural area, has a relatively high level of integrity for a county that has undergone rapid development. Additionally, county level policies support the maintenance of a core mixed agricultural area though some lands that are currently in agricultural use are likely to be converted to other uses as population growth continues (Chester County Planning Commission 2018c). These factors have important implications for the persistence of agricultural land uses and habitat availability for grassland birds.

Daniels (2004) described the need to protect a critical mass of farmland to maintain a sustainable agricultural industry. A county generally needs to maintain at least 40,000 hectares of farmland to maintain its agricultural economic base (T. Daniels, pers. comm.). In Chester County, there are 68,710 hectares in agriculture use, about 32,020 hectares of which are in the study area and represent the highest density of grass-cropland in the county. Ten percent of these grass-cropland patches are located in protective agricultural zoning though the level of protection varies with minimum acreage requirements ranging from 1 to 25 acres. Agricultural zoning below a 20 acre minimum lot size does not offer much protection for agricultural land uses and can, in fact, encourage low density sprawl.
In Chester County, the amount of grass-cropland area does not meet minimums described in the Bird Conservation Area (BCA) model for large-scale management of grassland birds (Sample and Mossman 1997), which requires a core area of about 800 hectares surrounded by a larger matrix of more than 4,000 hectares of agricultural land use. However, those standards were set for the Midwest and Great Plains where there are still vast grasslands. Preservation in the county has provided large protected areas, so grassland habitat is available. If the grass-cropland patches remain embedded in a landscape matrix that is structurally open (i.e., buffered from woodlands with limited development), it is possible to sustain small-scale grassland bird management in the study area (Davis 2004; Ribic et al. 2009).

In the absence of active bird management, there is a grassland bird guild in the agricultural belt of Chester County. Just having the land in agricultural production may be enough to support this bird community, but land protection is only the first phase of strategic preservation. The second phase involves land management, though this is more difficult with an easement designed to protect agricultural land because they are limited in their conservation value. Nevertheless, the next challenge in moving forward with active grassland bird management is developing more “bird friendly” land management.

Grassland bird management in Chester County may be best implemented within the context of the source-sink metapopulation model. If the landscape can be managed to provide high quality source areas supported by lower quality sinks, small populations of grassland birds can remain stable over the entire landscape. To do this may require changing our perception of the landscape. Grass-cropland patches in the county will take on greater conservation value if they are considered from the perspective of a bird that
perceives contiguous areas of grassland habitat as a single patch (though the structure may vary across the patch), rather than viewing the landscape as discrete patches bound by common management practices and human constructs (Ribic et al. 2009). If planning is based on less narrowly defined patch boundaries (i.e., parcels or smaller management units), the landscape becomes more fluid and offers greater opportunity for movement between source and sink habitats.

Managing for grassland birds will challenge current practices, but there are large biological and economic improvements that can be made through better spatial management (Polasky et al. 2008). Since land trusts and the Agricultural Land Preservation Board (ALPB) have different goals, land trusts could use different criteria for preservation that complement agricultural preservation. The Brandywine Conservancy, which holds about 40 percent of the conservation easements could be a partner in grassland bird management, working with the county to improve grassland bird management through a model in which the Conservancy holds easements on habitat sensitive land within the agricultural economic belt. In addition, though agricultural easements are usually explicitly designed to protect commercial farming interests, the ALPB does include management criteria, such as requiring a soil and water conservation plan. These criteria, which meet other state mandated goals (Department of Environmental Protection 2012), can be executed in a way to support grassland bird conservation. These might include requiring no-till farming, using conservation buffers designed for wildlife benefits, or installing site-level buffers that have landscape scale benefits (Heard et al. 2000; Henningsen and Best 2005).
For properties that have already been preserved, the predictive habitat occupancy maps produced through this study show areas where targeted management practices will be especially important because grassland species have been confirmed there or there is a high likelihood of species occupancy. Based on individual species preferences, much of the grass-cropland network in Chester County would not provide primary habitat. Therefore, the underlying goal would be to prioritize lands to be incorporated into the existing grass-cropland network to optimize the spatial configuration of the network and to develop best management practices that support grassland bird species and encourage their persistence through use of alternative habitats. Figure 21 provides a snapshot of the study area. Using Bobolinks as an example, there are three areas in the landscape where the probability of Bobolink occupancy is greater than 70 percent (see Figure 22a), between 40 and 60 percent (see Figure 22b), and below 30 percent (see Figure 22c). This kind of fine grain assessment could raise awareness of a property’s value for grassland birds and could help guide land management at the parcel level.
The findings from this research point to several factors that could reinforce grassland bird conservation within the study area. Though there are no growth boundaries in the county, there are de facto growth boundaries along the border of the study area where preserved land abuts the boundaries of growth areas. This permanent demarcation will not allow growth to spill into the rural resource area. However, there are areas that have not been preserved and are susceptible to development—in the southwest there are protected agricultural lands that could be separated from the contiguous block of preserved land because there are a number of unpreserved parcels between them. Though preservation has been targeted for this area, it may not be enough to avoid isolation of
those agricultural lands. This area would benefit from restrictive agricultural zoning, which would provide a physical and temporal buffer until the land can be preserved if the ultimate goal is to maintain the land in agriculture.

Though development is to be limited in the rural resource areas, there are over 30 municipalities that are fully or partially contained in these areas. For the county to meet its stated conservation planning goals, individual municipality policies will need to align with *Landscapes3*, and those municipalities with a high proportion of farmland should be encouraged to adopt strong agricultural zoning. The probability models suggest agricultural zoning, or, at least, the aggregation of land it supports, benefits grassland bird species and is a factor related to habitat suitability. Land use regulations within rural municipalities that protect agricultural land uses and limit development could have a positive impact on conservation planning and could be used to keep the landscape open and implement a variety of sustainability goals for rural resource areas.

However, because it is not possible to preserve all susceptible land and there may not be the political will to support agricultural zoning, other inducements could be used to keep land in agricultural uses. One incentive is to maintain unprotected agricultural land is Chester County’s Clean and Green Program. The program offers preferential tax assessment to reduce the property tax burden on farmers who have a minimum of 10 acres (4 ha) or have an annual gross agricultural income of $2,000 (“Act 319 - ‘Clean & Green’” 2019).\(^{24}\)

\(^{24}\) Land assessment is based on the agricultural value of the land and soil productivity rather than its potential development value if it is located in a rural or residential zone. To protect against abuse of preferential assessment, if a parcel is enrolled in the program and is then developed, the landowner must pay roll-back taxes.
It may also be possible to enlist the help of the farmers in conserving grassland birds. In Chester County, current haying regimes have the potential to be altered to enable grassland birds to have higher nesting success in artificial hayfields. Horse hay is cut early, beginning between May 15 and May 20 and is finished by the first or second week in June. The remainder of the hay is mulch hay used by the large mushroom farming industry in the county. Horse hay is distinguished by its appearance rather than specific measures of crude protein (i.e., green and not stalky), and it commands $625 more per hectare than mulch hay. However, because of the effort required to cut, dry, and bale a field, it is only possible to have a portion of the fields used for horse hay. Most of the hay in the study area (~80 percent) is mulch hay and can be cut later because it does not need to meet standards of quality, and leaving it to cure in the fields is more beneficial for mushroom growing (J. Hicks, pers. comm.). With the two types of hay, it would be possible to establish an early cut on some fields that would enable birds to re-nest if they are disturbed and a late cut that would occur after birds have fledged. This partitioning of the haying schedule would maximize nesting opportunities on both types of fields.

There are only three large farming operations that do the majority of the mowing in the study area. They lease the land and develop the schedule, so they would need to be included in the planning. Based on personal interviews with some of the farmers, there is a willingness to engage in a more formalized program. Landowners would also need to be included, because some of the mid-season haying is done for aesthetic reasons, and, with more education, landowners might be willing to delay cutting to benefit the birds.

Nevertheless, the county’s population is expected to grow to by over 140,000 by 2045. Accommodating this growth while maintaining grassland bird populations will be a
challenge. To encourage more compact growth, Chester County’s *Landscapes3* plan calls for coordinated planning of water, sewer, and other infrastructure with new development, such that areas that already have sewer and water are prioritized for development (Chester County Planning Commission 2018c). Since public infrastructure and facilities are catalysts for land development (Brody and Highfield 2005), requiring adequate public facilities will help direct development to growth areas and away from areas targeted for preservation. Near the study area, incentives could encourage infill or redevelopment of under-used parcels in the urban and older suburban areas of Coatesville and promote higher population densities in designated growth areas away from agricultural and rural resource areas. Overall, Chester County has a strong plan in place for both growth and preservation; however, it will take a coordinated effort among the many municipalities in the study area to implement effective zoning and other growth management tools that will support active grassland bird management. Nevertheless, given the existing conditions, a collaborative effort to extend and strengthen grassland bird conservation could be viable in Chester County.

**Conclusion**

There is an inherent tension in attempting to preserve land for biodiversity conservation: there is not enough money to purchase enough habitat for all species to exist in “wilderness” conditions, but the more cost-effective approach of enlisting private lands may not go far enough in attaining conservation outcomes because human land use can threaten ecosystem health. In the U.S., the challenge is to prioritize habitat most vulnerable to conversion or degradation and to develop a robust habitat network.
throughout landscapes that are often characterized by wasteful land use and zoned for sprawl. Traditionally, planners have not used knowledge about ecological processes to inform spatial planning (Opdam et al. 2001). To be more effective, planners will need to take a strategic, science-based approach that makes site selection for habitat preservation more efficient and more likely to address the conservation needs of vulnerable species. Exploring the relationship between landscapes and biological data provides a valuable tool for conservation planning. With limited funding for land preservation, conservation must be done wisely. Therefore, when preserving land, planners will need to use the available tools to build a spatially efficient reserve network that provides high-quality habitat and buffering private lands that further biodiversity goals.

My research supports this goal theoretically and provides the opportunity to pursue real conservation outcomes for grassland birds. The predictive models I developed for habitat occupancy could inform local planning decisions and enable more strategic conservation planning that targets habitat patches that have a high probability of providing grassland bird habitat. These ecologically-based tools were developed at a scale that could address the needs of smaller land preservation organizations seeking to augment regional land preservation efforts and could be scaled up to incorporate a wider area. These tools would also enable planners to identify individual sites that are likely to support multiple grassland species to determine how well these sites are protected.

In our agricultural lands, grassland bird populations are under pressure from land conversion and intensification, but Chester County provides a test case for maintaining a small, healthy and persistent population. My dissertation research provides a pathway to achieve these biological outcomes in Chester County. The census results show small
populations of grassland birds are in Chester County. The landscape diagnosis, which was not meant to provide definitive results, but, rather, to explore how well current planning initiatives would align with goals for a multifunctional landscape, shows land in the study area is aggregated and protected in patterns that make it possible to sustain small-scale grassland bird communities that will have some overlap but will also have areas that are only suitable to a subset of the focal species. These findings provide the opportunity to use the existing landscape and make it more suitable for grassland birds through small-scale and targeted management practices.

Conservation planning has played an important role in preserving biodiversity and protecting ecosystems, and our need for it is growing as we plan for the impacts of climate change. Over the last 40 years, changing trends among birds, which are sensitive to environmental change, have coincided with temperature increases (National Audubon Society 2009). Among landbird species reported on in *Birds and Climate Change: Ecological Disruption in Motion*, 64 percent showed significant northward movement from 1969 to 2009. However, grassland birds were among the few avian guilds that did not exhibit a northern shift in range during that period (National Audubon Society 2009). If this trend continues in which grassland bird ranges remain stable, there will need to be greater emphasis on planning for these species in their current range. This means there will need to be adequate habitat of sufficient quality and connectivity available to support healthy populations of birds, and in areas where there is increasing development pressure on habitat, there will need to be greater balance between economic and ecological considerations. But getting it right could make the difference in the persistence of these
birds whose burbling, buzzing, whistling songs float up from fields in summer to remind us of a cycle of renewal.
APPENDIX

Assessment Framework for Chester County Land Use and Conservation Planning

Where Are We Now?

1. Characterize the county’s current land use patterns and conservation efforts. Have wasteful development patterns peaked, or are the trends of mid-1990s continuing (when Chester County’s first comprehensive plan, Landscapes, was adopted)?
   a. What is the evidence for this?
   b. What are the factors that have contributed to the current conditions?

2. What historical trends have informed past and present conservation actions in the county?
   a. How are these trends affecting the organization’s approach to conservation?
   b. What are the greatest threats to habitat/agricultural lands? How is the organization targeting these threats through land use and management?
   c. Are there areas of the county where these threats are more pronounced?

3. Has your organization identified gaps in the conservation network?

![Map showing land preservation efforts in Chester County]
Where Do We Want To Be?

4. What land-use patterns does your organization want to see in the next 20 to 25 years?
   a. Will it be important to protect certain land uses over others?
   b. Do those uses change depending on the location within the county?
   c. Will the organization concentrate its efforts in the Rural Resource Areas of the county or take a more uniform approach to undeveloped lands throughout the county?

5. Is your organization focused on maintaining pockets where historical land uses prevail or developing a green infrastructure model in which an interconnected network of preserved land provides the skeleton for the county?

6. How does your organization determine land conservation targets?
   a. Are the organization’s conservation efforts guided by an over-arching focus (e.g., preserve habitat, maintain critical mass of farmland, restore riparian lands and immediate surrounding areas, provide recreational opportunities)?

7. Are the organization’s activities led by a plan that defines goals and actions for implementing the plan?
   a. Is this a formal document that is available to the public?
   b. To what degree does the plan focus on ecological connectivity (e.g., hubs, corridors, and buffering lands)?
   c. To what degree does the plan focus on providing open space for human benefits?
8. Which type of network is most likely to occur in the county?
   a. functional site – conserving a small number of ecosystems or species in a smaller area
   b. functional landscape – conserving many ecosystems and species, but not on a regional scale
   c. functional network – conserving an integrated set of sites and landscapes to conserve regional-scale species

9. What kinds of actions/programs does your organization need to put in place or change to make that landscape a reality?
   a. How do you use specific mechanisms/tools for land protection to achieve your organization’s goals (i.e., land acquisition, conservation easements, joint partnerships, other)?
   b. How does the organization maintain consistency when applying criteria for land acquisition, so each acquisition is part of a larger vision/strategy rather than the result of opportunity?
What Do We Need?

10. What data are used to prioritize lands for preservation and to assess lands that are up for preservation?
   a. Is the organization’s planning approach flexible enough to allow for the incorporation of new data as they become available?

11. Does your organization have a methodology for quantitatively ranking conservation opportunities?
   a. What factors are used to develop the ranking system (e.g., specific conservation outcomes, overall goals to build a connected conservation network, or other important considerations)?
   b. Can this methodology help guide implementation efforts at a range of scales (from coarse-scale to site-level assessments)?

12. At what scale do you focus your efforts?
   a. local scale – site-level
   b. intermediate scale – multiple small-patch ecosystems
   c. coarse scale – dominant (or historically dominant) landscape components in which large- and small-patch ecosystems are embedded (e.g., deciduous forest that includes wetlands, aquatic ecosystem, etc.)
   d. some combination of scales

[Diagram of Biodiversity and Scale]

From Biodiversity Conservation at Multiple Scales: Functional Sites, Landscapes, and Networks (Potani et al. 2000)
How Do We Go About It?

13. How would your organization rate the following conservation priorities? How are these objectives reflected in the composition of land holdings?

<table>
<thead>
<tr>
<th>Preservation Objectives</th>
<th>Priority H/M/L</th>
<th>% of Holdings</th>
</tr>
</thead>
<tbody>
<tr>
<td>Targeting biologically important lands (e.g., to protect rare/threatened species or to maintain certain ecosystem functions and processes)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Contributing to ecological integrity</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Preserving land threatened by encroaching development (e.g., remnants of historical landscape or pockets of green space)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maintaining lands primarily for agriculture with some value as habitat or for contribution to ecosystem integrity</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Retaining land that conserves open space values</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

14. How did your organization define what types of properties it would seek for conservation?
   a. What provisions has the organization made to achieve these priorities?
   b. Does the organization have written criteria for what defines specific conservation values and standards for what constitutes an excellent example of those values?
   c. If specific species/ecological functions are being targeted, what are they?
   d. Do funding sources affect the types of land uses/properties that are targeted?

15. Has the organization coordinated its goals with state or local growth management efforts or with other conservation organizations?
   a. How much does the organization consider local zoning or other land-use regulations that might help or hinder conservation when evaluating which properties to preserve?
   b. Does the organization’s plan call for coordination among municipalities to address issues that extend beyond jurisdictional boundaries?
   c. Does the organization work with municipalities to take a more regional approach to land conservation?
16. Has your organization identified opportunities for development within the context of the conservation network, so new growth can be directed to the most suitable areas?

17. What management practices does the organization require?
   a. How do you define management goals to maintain conservation value over time?
   b. What is the purpose of those practices?
   c. How much can be required before meeting with resistance?

What Results Have We Achieved?

18. Have the organization’s land preservation goals changed over time? How has this affected the approach the organization takes?

19. Are there specific properties or areas in the county that have been preserved but have failed to live up to the needs they were meant to fulfill? In other words, are there preserved properties/areas you would no longer target in 2015?

20. What means of assessment are in place to judge current and future activity?

21. What proportion of the new land deals that are pursued are chosen strategically to build onto existing landscape spatial patterns to make better use of the existing network?

22. How does the organization assess the outcomes of individual transactions and their impacts at the landscape scale?
   a. acreage protected
   b. proximity to other preserved lands (connectivity)
   c. ecological significance
   d. quantitative changes that coincide with conservation
   e. protection of “critical landscape”


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