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Greenspace Conservation Planning Framework for Urban Regions Based on a Forest Bird-Habitat Relationship Study and the Resilience Thinking

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GREENSPACE CONSERVATION PLANNING FRAMEWORK FOR URBAN REGIONS BASED ON A FOREST BIRD-HABITAT RELATIONSHIP STUDY AND THE RESILIENCE THINKING

A Dissertation Presented

by

Sadahisa Kato

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ABSTRACT

GREENSPACE CONSERVATION PLANNING FRAMEWORK FOR URBAN REGIONS BASED ON A FOREST BIRD-HABITAT RELATIONSHIP STUDY AND THE RESILIENCE THINKING

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The research involves first conducting a “case study” of ecological data and applying the results, together with the resilience concept, to the development of a greenspace conservation planning framework for urban regions. The first part of the research investigates the relationship between forest bird abundance and the surrounding landscape characteristics, especially, forest area and its spatial configuration in urban regions at multiple scales. The results are similar for simple and multiple regression analyses across three scales. The percentage of forest cover in a landscape is positively correlated with bird abundance with some thresholds. Overall, the percentage of forest cover in the landscape, contrast-weighted forest edge density, and the similarity of land cover types to forest cover are identified as important for the conservation of the target bird species. The study points to the importance of species-specific habitat requirements even for species with similar life history traits and of maintaining some forest edges and/or edge contrast. The second part of the research involves the development of a landscape planning meta-model and its conceptual application to greenspace conservation

planning, integrating the results of the first part. Administrative and planning units are recognized to exist in a nested hierarchy of neighborhood, city, and urban region, just as biodiversity can be conceived in a nested hierarchical organization of genes, populations/species, communities/ecosystems, and landscapes. Resilience thinking, especially the panarchy concept, provides a scientific basis and a metaphorical framework to develop the meta-model, integrating a proposed landscape planning "best practice" model at each planning scale. Ecological concepts such as response and functional diversity, redundancy, and connectivity across scales are identified as key concepts for conserving and increasing biodiversity and the resilience of an urban region. These concepts are then used in the meta-model to develop the greenspace conservation planning framework. Ecological processes such as pollination and dispersal, as well as social memory and bottom-up social movements—small changes collectively making a large impact at the broader scales as well as these incremental changes gaining momentum as they cascade across scales—are identified as cross-scale processes and dynamics that connect various planning scales in the meta-model.

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CHAPTER 1

INTRODUCTION

1.1 Introduction to the Research Problem

The loss of biodiversity is one of the most critical and persistent environmental problems in the world over the last 20 years as recognized by a recent United Nation's report (i.e., Global Environment Outlook, or GEO-4) (United Nations Environment Programme [UNEP] 2007). The current rate of species extinction is a hundred times faster than the rate shown in the fossil record (Millennium Ecosystem Assessment [MA] 2005, UNEP 2007). Of the major vertebrate groups that have been comprehensively assessed, over 30 percent of amphibians, 23 percent of mammals, and 12 percent of birds are threatened (MA 2005, UNEP 2007). Biodiversity suffers, for example, from urban (suburban) sprawl and growing demand for food, leading to either intensified agriculture (using more chemicals, energy and water, and more efficient animal breeds and crops) or by cultivating more land (MA 2005). Besides the argument for its intrinsic value and humans' ethical responsibility to protect it, biodiversity needs to be protected because it plays multiple roles in the daily lives of people through the provisioning of ecosystem services (McNeeley et al. 1990, Peck 1998, MA 2005, Groom et al. 2006). Humans rely on ecosystem services originating from biodiversity, including food, fuel, fiber, medicines, air, and soil (McNeeley et al. 1990, MA 2005, Ahern et al. 2006). Biodiversity loss and ecosystem service changes are an interrelated issue, affecting many fundamental aspects of human well-being (Peck 1998, MA 2005).

A broad consensus in the scientific and policy communities exists regarding the causes of biodiversity loss. Habitat loss, fragmentation, and degradation are the major causes of global biological diversity decline (Noss 1991, Fahrig 1997, Peck 1998, Wilcove et al. 1998, Pullin 2002, Groom et al. 2006). The decline can be slowed, stabilized, or in some areas even reversed by (1) policies, strategies, and management that (a) protect, restore, and create habitats, and (b) mitigate habitat loss and fragmentation and by (2) "smart" plans/designs that can accommodate both plants', animals', and people's needs (MA 2005, Ahern et al. 2006, Collinge 2009). Landscape ecological planning, which integrates landscape ecology into landscape planning and focuses on people's interaction with nature, can arguably contribute to developing plans and designs that can accommodate these needs (Cook and van Lier 1994, Langevelde 1994, Gutzwiller 2002, Ahern et al. 2006, Noss and Daly 2006). Because landscape ecology deals with the relationship between landscape structure and ecological processes at various spatial and temporal scales (Risser et al. 1984, Forman and Godron 1986, Turner 1989, 2005, Turner et al. 2001, Fortin and Agrawal 2005, Farina 2006), some of the principles and theories (e.g., the aggregate-with-outliers principle [Forman 1995] and the concept of connectivity) of landscape ecology could inform landscape planning in such a way to lessen the effects of, especially, habitat loss and fragmentation and to achieve a spatial configuration of land use that provides habitat for species and accommodates room for development.

This research focuses on analyzing, planning, and designing the spatial configuration of land uses/covers at broad scales—a fundamental aspect of any environment, including a built environment. I argue that for any given density of people or protected areas, there are better or worse spatial configurations, and these can be informed by theories and principles from landscape ecology (Forman 1995). Thus, my argument can be understood as “smart” land use (more holistic approach to landscape planning, integrating people and natural systems) vs. creating habitat by removing or limiting people (more traditional approach to nature protection such as protected reserves and national parks in remote areas). The concept of holism is based on the notion that a whole (such as a landscape) is more than the sum of its parts (Zonneveld 1990, 1995). When the concept is applied to the study, planning, and management of a landscape, the focus should be on the interactions among its components or between human and natural systems, not knowing every small detail about all the components (Zonneveld 1990, 1995, Ndubisi 2002a). Its utility at an operational level, however, is a topic of much discussion (Ndubisi 2002a).

The big issue driving this research is the loss and fragmentation of wildlife habitat and its effects on native species. I have chosen forest birds as the focal species, species that are arguably critical for, and indicators of, maintaining ecologically healthy conditions (Benedict and McMahon 2006). There are practical reasons why birds are often used in landscape ecological studies. Some birds are fairly conspicuous and thus can be easily observed. Birds can be identified by their characteristic songs as well. They are ubiquitous and their habitat requirements are relatively well-studied (Morrison 1986, van Dorp and Opdam 1987, Harms and Opdam 1990). Birds have been used as the indicators of changes in habitat amount, spatial configuration (e.g., connectivity), and quality (Whitcomb et al. 1981, Morrison 1986, Bolger et al. 1997, Rosenberg et al. 1999, Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto et al. 2005, Sandström et al. 2006). Because birds fly, they are inherently adaptable to higher levels of fragmentation (e.g., in urban areas) than other species that require physical, terrestrial or aquatic linkages.

Researchers have associated the distribution and abundance of birds with habitat variables (e.g., habitat composition, configuration, and quality) to create potential habitat maps of the species targeted for conservation and to determine the habitat factors that are important for the conservation of the bird species of interest (Whitcomb et al. 1981, Morrison 1986, Bolger et al. 1997). Forest birds, in particular, have been used as a response variable to measure the effect of habitat fragmentation in general due to urbanization and the conversion of forests to agricultural lands (e.g., Rosenberg et al. 1999, Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto et al. 2005, Sandström et al. 2006). These studies used forest bird species richness (i.e., the number of species) and/or the presence/absence of individual species as the indicators of the quality of urban green spaces (e.g., the composition of vegetation, the size and configuration of urban parks), or as the response variables to the composition and configuration of forest patches.

Some studies focused on the spatial configuration of forest patches. For example, Rosenberg et al. (1999) used Tanagers (*Piranga* spp.) and Fernández-Juricic (2004) used forest passerines as the indicator of forest fragmentation in general based on these birds' life history characteristics. Because forest-interior birds (and some ground-nesting species) are threatened by fragmentation (Marzluff 2001)—for example, susceptible to increased nest predation and brood parasitism by Brown-headed Cowbirds (*Molothrus ater*) (Robinson 1992), their abundance and occurrence can be used as the indicator of forest fragmentation (Rosenberg et al. 1999, Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto et al. 2005, Sandström et al. 2006).

As forests become fragmented, the interior area of a forest patch, habitat of the selected forest birds (Roth et al. 1996, Gough 2007), decreases in size. The point is that fragmentation per se increases the amount of edge (Hansbauer et al. 2008). The amount of edge is influenced by the shape of a patch: the more compact a patch is, the less the amount of edge is as compared to more convoluted and elongated patches (Forman 1995, Ewers and Didham 2006). Vegetation composition and structure differ from the edge of the forest to the interior affected by light availability, humidity, air temperature, wind, etc. (Forman and Godron 1986, Watson et al. 2004). The edge effect refers to this differential species composition and abundance found in the edge of a patch as compared to those found in the interior of a patch due to the difference in the microenvironment (Forman and Godron 1986). The edge effects have been reported to extend at least 100 to 250 m from the edge and in some cases up to 1 km (Piper and Catterall 2006, Twedt et al. 2006). Edge species are found only or primarily near the perimeter of a landscape element; interior species are found only or primarily away from the perimeter of a landscape element (Forman and Godron 1986). Therefore, species that specialize in interior habitat are negatively affected by the increase in edge habitat. The effects of increased edge on forest-interior birds include lower species richness, Shannon diversity, and abundance (Germaine et al. 1997, Laiolo and Rolando 2005, Gentry et al. 2006). Increased nest predation and brood parasitism by Brown-headed Cowbirds, caused by the increase in forest edge, are said to be one of the mechanisms that reduces the abundance of the selected forest breeding birds, which are neotropical migrants (Yahner and Scott 1988, Gustafson and Crow 1994, Robinson et al. 1995).

There are relatively few bird-habitat relationship studies in urban areas (e.g., Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto, et al. 2005, Sandström et al. 2006) and/or at a broad spatial scale such as a regional (landscape) scale (e.g., Whitcomb et al. 1981, Askins et al. 1987, Flather and Sauer 1996, Bolger et al., 1997, Boulinier et al. 2001, Donovan and Flather 2002, Vance et al. 2003, Pidgeon et al. 2007), as compared to the patch scale. Urban studies are few because traditionally ecologists have worked in pristine environments away from human settlements (Collins et al. 2000). Regional scale studies are few because of various limitations including time, budget, and personnel. Urban regions are where most people live in the United States (U.S.) (Hobbs et al. 2002) and often coincide with the areas of high biodiversity conservation priority (Groves et al. 2000, Balmford et al. 2001, Araújo 2003). The urban regional scale investigation is arguably necessary to develop a conservation plan that covers a broad area where the persistence of regional populations of birds can be ensured because: (1) some bird species (e.g., predatory species) require a large territory or home range (Keitt et al. 1997, Thompson and McGarigal 2002); (2) some birds display metapopulation dynamics in an increasingly fragmented landscape (Opdam 1991, Opdam et al. 1995); (3) some birds, such as forest birds, have a long dispersal range and neotropical migration (Friesen et al. 1995, Robinson et al. 1995, Donovan and Flather 2002); and (4) opportunities exist to develop “smartly,” lessening the impact of land use on biodiversity, mitigating the loss, and even creating new habitat. Therefore, more research is needed to investigate the bird-habitat relationship at the scale of a large urban/metropolitan region as a whole or even across multiple urban regions. Using forest-interior bird species as the indicator of broader biodiversity, an urban regional-scale study of the bird-habitat relationship would contribute to developing a regional goal for biodiversity conservation and advance landscape ecological planning that would support biodiversity in a broader urban/metropolitan region.

When considering the effects of habitat loss and fragmentation on the abundance and occurrence of forest birds in large urban regions, the critical threshold of habitat connectivity (With and Crist 1995, Wiens et al. 1997, Turner et al. 2001) is an important concept that affects the dispersal/movement of forest birds and therefore the persistence of regional forest bird populations as potential metapopulations (Opdam et al. 1995). The critical threshold of habitat connectivity is the amount (percentage) of habitat in a landscape below which the habitat becomes functionally disconnected for an organism moving across the landscape (With and Crist 1995, Fahrig 2001, Turner et al. 2001). "In landscape ecology, substantial theoretical progress has been made in understanding how critical threshold levels of habitat loss may result in sudden changes in landscape connectivity to animal movement. Empirical evidence for such thresholds in real systems, however, remains scarce" (Olden 2007). Although abrupt changes (i.e., thresholds) have been precisely defined in simulated landscapes (e.g., Gardner et al. 1987, With and King 1997, With et al. 1997, Fahrig 2001), such changes in the structure of real landscapes are not well understood. Thus, the threshold concept is an important theory to be examined in the landscape ecological data analysis, and in the context of forest birds, specifically.

Simulation models predict sudden changes in species occupancy and population persistence at the critical threshold of landscape connectivity (Gardner et al. 1987, With and Crist 1995, Fahrig 2001). This research adds to few existing empirical studies (Andrén 1994, Wiens et al. 1997, McIntyre and Wiens 1999) that tested the predictions of simulation studies by comparing multiple urban regions with different percentages of tree cover and connectivity with respect to the abundance of the individuals of the selected forest bird species (see section 3.2.3). By studying the landscape surrounding the bird survey routes in urban regions across the eastern U.S., this research covers a wide gradient of forest amount and spatial configuration. The research also provides a good opportunity for testing an interesting finding of earlier simulation and empirical studies that found a stronger influence of forest spatial configuration on the abundance and occurrence of forest birds when the amount of forest in the landscape is low (Cooper and Walters 2002, Flather and Bevers 2002, Betts et al. 2006b).

In summary, declining biodiversity is a global concern and landscape ecological planning can contribute to protect and in some cases even increase biodiversity. Forest-interior birds can be an indicator of the loss and fragmentation of forest habitat, often caused by suburban sprawl and conversion to agricultural lands. Forest birds can arguably be an indicator of other forest-dependent fauna and flora, and of associated ecosystem functions such as water filtration, preventing soil erosion, air purification, carbon sequestration, and cultural, recreational, and economic benefits that healthy forest ecosystems can provide. Although there are many studies that investigated the relationship between the structure of green spaces (e.g., their size, shape, spatial configuration, and vegetation composition) and forest birds within city boundaries (Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto et al. 2005, Sandström et al., 2006) or at a site-scale (based on each forest patch), few have examined this relationship at the scale of a larger urban/metropolitan region as a whole or across multiple urban regions (Whitcomb et al. 1981, Askins et al. 1987, Flather and Sauer 1996, Bolger et al., 1997, Boulinier et al. 2001, Donovan and Flather 2002, Vance et al. 2003, Pidgeon et al. 2007). Therefore, more research is needed to investigate the bird-habitat relationship in a more regional/metropolitan scope, examining the spatial composition and configuration, and thresholds of forest cover to help establish a regional goal for biodiversity conservation along with other compatible planning goals and to advance landscape ecological planning that supports biodiversity in a broader urban region.

1.2 Statement of Purpose

In order to examine the relationship between forest bird abundance and their habitat structure at a broad, regional, metropolitan scale, I propose to investigate the relationship between the percentage of forest cover and its spatial configuration, especially, connectivity, and the number of individuals of the selected forest-interior breeding bird species in major urban regions across the eastern U.S. Urban region is defined as a spatial/geographical entity that is composed of interacting abiotic, biotic, and cultural resources, and can be composed of multiple jurisdictions (e.g., a core city or cities and its surrounding suburbs that have strong social and economic ties to the core, as measured by, for example, the amount of public transportation and the percentage of people commuting to the urban core) (McDonnell and Pickett 1990, Forman 1995, 2008, Medley et al. 1995, Foresman et al. 1997, Steiner 2002a). (Urban regions in this study are defined by Metropolitan Statistical Areas [U.S. Office of Management and Budget 2000] as described in chapter 3.) The abundance of forest breeding birds can be considered as an indicator of forest loss and fragmentation, and broadly, of forest ecological functions (e.g., water holding, gradual release of water to underground aquifers, air purification, providing habitats for other forest fauna and flora) as influenced by forest composition and configuration (Forman and Godron 1986, Rosenberg et al.1999, Marzluff 2001, Fernández-Juricic 2004). By focusing on the bird-habitat relationship in urban regions and forest bird threshold response to the percentage of forest cover, the comparative observational study will add to the few prior empirical studies that have tested the existence of threshold of habitat loss. Further, the results can have implications for planning appropriate amount of forest cover and the degree of connectivity of forest cover in urban regions for the focal forest breeding bird species and other forest-associated species/functions that the forest birds are assumed to represent.

By investigating the bird-habitat relationship across multiple urban regions, useful insights can be drawn for the maintenance of forest bird populations in these urban regions across the eastern U.S. Conservation planning recommendations can be applied to urban regions with a similar percentage of forest cover and connectivity as compared to the results of earlier studies conducted either within specific urban areas, often within the administrative boundaries of cities/towns, or in relatively pristine environments (e.g. National Parks). Urban regions (i.e., my study areas), just as any landscape, have multiple spatial scales (e.g., neighborhood, city, urban region) as a potential planning unit. This multi-scale nature of the study areas creates the need to describe/analyze and pose planning recommendations for these geographical areas in a nested or hierarchical sense. For example, there may be different thresholds at different scales, and the planning and policy recommendations, and implementation tools and strategies will surely differ from the neighborhood scale to the urban regional scale. Since urban regions include the suburbs of a core city (or cities) and the suburbs (i.e., the urban fringe areas) are under increasingly strong development pressure—an issue which many communities that surround large cities face today, the management and planning of these suburban landscapes as part of a larger urban region has implications for not only conserving biodiversity but also the quality of the lives of the residents.

I am also interested in how landscape ecology concepts and principles can inform better land use planning, one that would integrate the needs of both human and non-human species and ultimately, one that would increase the sustainability of landscapes. The purpose of the second part of the research is, based on the comparative observational study of the forest bird-habitat relationship, to develop and apply a landscape planning framework for conservation of regional biodiversity. The planning framework in practice would result in a conservation/land-use plan that may increase the percentage and connectivity of forest cover, and by doing so, would contribute to the conservation of not only the selected forest birds that have shown a declining trend but also broader biodiversity of which these birds are part, and other associated ecological, social, economic, aesthetic, educational functions that tree covered (forested) areas can reasonably provide.

1.3 Research Hypotheses and Questions

The research has two major parts. One is the analysis of ecological/geographical/spatial data: the route-level, multi-scale analysis of forest bird abundance with regard to forest loss and fragmentation in urban regions across the eastern U.S. The other is the development of an original landscape planning framework for greenspace conservation planning in urban regions based on the resilience concept.

1.3.1 Landscape Ecological Data Analysis

The hypothesis is that both the percentage of forest cover and the degree of its connectivity in the vicinity where the selected forest breeding bird species are observed are positively correlated with the number of individuals of the selected forest bird species in urban regions across the eastern U.S. In other words, (1) the higher the percentage of forest cover is, the more abundant the selected forest birds become and (2) the more connected forest cover is, the higher the number of the individuals of the selected forest birds is.

In addition to the hypothesis to be tested, the following research questions are investigated:

- To what degree do the percentage and connectivity of forest cover affect the selected forest breeding bird abundance? Can increased connectivity compensate for reduced forest area?
- Do the selected birds exhibit a threshold response to the percentage and/or connectivity of forest cover? If so, what is the threshold for percent forest cover and for forest cover connectivity?
- Is the spatial configuration (e.g., patch shape, patch isolation, proximity to edge, connectivity) of forest cover important for the abundance of the forest birds, especially near the identified threshold?

- How do these two factors—percent forest cover (composition) and forest cover connectivity (configuration)—interact to predict the abundance of forest birds?
- How do these factors vary when measured at different spatial scales?
- What functional connectivity measures are available to predict the forest breeding bird abundance?
- What is the optimum percentage of forest cover for the selected forest birds?
- What is the range of connectivity that best supports the forest breeding birds?
- Is there a generalizable relationship between the appropriate percentage of and degree of connectivity of forest cover and the breeding bird abundance across different geographic/climatic regions in the eastern U.S.?
- What land cover/use type including forest cover is the best predictor of the forest bird abundance?
- What would be a reasonable goal for the percent forest cover and its connectivity to support the selected forest birds in the urban regions across the eastern U.S.?

One of the two major objectives of the landscape ecological data analysis is to determine if there is a threshold percentage of forest cover below which the abundance of the selected forest birds declines significantly, suggesting the minimum percent forest cover required to maintain the populations of these forest bird species. The other objective is to determine if the spatial configuration (e.g., connectivity) of forest cover, independent of area, is important at all for the abundance of the forest birds. Further, the study is intended to evaluate the relative importance of the amount and configuration of forest cover near the amount threshold (if identified)—alternatively, how (or if) they interact to influence the abundance of the individuals of the selected species of forest birds. This interaction between the amount and spatial arrangement of forest cover, if found, may generate an interesting hypothesis that is of conservation planning significance: if forest covers are connected, a smaller percentage of forest cover may be necessary to sustain the forest bird populations.

1.3.2 Development of a landscape planning framework

The second part of the research will involve mostly literature review on landscape ecology principles and theories and on landscape (ecological) planning strategies and concepts for the development of an operational landscape planning framework for biodiversity conservation as the central planning goal in urban regional planning. The main research question is: How can planning and design cultivate or improve the capacity of an urban region to provide ecosystem services over time in the context of change? I will argue that response/functional diversity, redundancy, and connectivity across scales are key to the resilience of a social-ecological system and the sustained provision of ecosystem processes and services. Resilience refers to "the capacity (or ability) of a system to absorb disturbance and still retain its basic function and structure" (Walker and Salt 2006, p. xiii). (See chapter 4 for more complete discussion on resilience and other related concepts such as adaptive cycle and panarchy.) Although there is a growing recognition that these ecological concepts are key to the maintenance of ecosystem functions over time, a link to planning and design application has not been strongly established. Then the question becomes: How can the concepts of response and functional diversity, redundancy, and connectivity across scales be translated to landscape planning and design—specifically, greenspace conservation planning in urban regions? I will argue that response/functional diversity and cross-scale connectivity are the aspects of a social-ecological system which planning and design can intentionally create, protect, or restore in a conservation planning framework, and which enable it to maintain its resilience.

Building on several recent general landscape planning models (i.e., Steinitz 1990, Steiner 1991, Ahern 1999, Leitão et al. 2006, Kato and Ahern 2008), I will develop a landscape planning "best practice" model, which will be integrated, with resilience thinking, into a landscape planning meta-model. Then, I will apply the meta-model in developing a greenspace conservation planning framework for urban regions for the planning goal of conserving regional populations of forest birds and other associated ecosystem services that forested areas can provide. The results of the forest bird abundance and habitat relationship study will be used in the development of the greenspace conservation planning framework.

The following questions will help me develop the meta-model and apply it to the general topic of greenspace conservation planning in urban regions:

- What planning strategies and policies can be applied to protect or restore the amount and spatial configuration of forest patches that support the selected forest bird species?

- What can be learned from prior landscape (land-use) plans with similar/related goals such as biodiversity conservation?
- How do plans differ with respect to scale (e.g., site, neighborhood, watershed, city, and urban region)?
- What "collateral" functions and benefits (e.g., recreation, water quality, and cultural landscape protection) can be expected to be associated with these plans/policies to protect certain species?

1.4 Assumptions and Limitations of the Study

1.4.1 Grain

Grain is defined as the minimum resolution of the data that is assumed to be homogeneous (Turner et al. 2001). The resolution of the land cover data used in this research is 30 m with the minimum mapping unit being 0.4 ha (1 acre) (Homer et al. 2007). Therefore, accuracy is limited to this resolution. It is assumed that this resolution corresponds with the perception of the target bird species; the birds are assumed to be able to grasp and respond to the features up to 30 m at one time. Although I could not find a study that tested this assumption, it seems reasonable compared to the perception of less vagile species such as beetles and amphibians.

1.4.2 Forest Cover as Habitat for Forest Breeding Birds

The original deciduous, evergreen, and mixed forest cover classes in the land cover map (see data source) are aggregated into a "forest" cover class and this forest cover class is assumed to be the habitat of the selected forest breeding bird species. This generalization and simplification is necessary to focus on the broad-scale pattern of the relative bird abundance and the habitat (i.e., forest) relationship. Other studies that combined similar broad-scale data (e.g., the North American Breeding Bird Survey and the U.S. Geological Survey land use/cover map) also aggregated different forest types into one forest cover type (Harms and Opdam 1990, Flather and Sauer 1996, Vance et al. 2003). For example, Harms and Opdam (1990) investigated woodland habitat for forest birds in the Randstad area in the Netherlands. They treated all forests equally for their habitat value. They assumed that all forests would develop into mature mixed deciduous forests, suitable habitat for any forest bird species in the region. Vance et al. (2003) aggregated coniferous, deciduous and mixed forest covers into one forest cover type, and compared this to all other land cover types—the approach I will take as well.

Limitations of the aggregation include over- or under-estimation of true habitat amount and assumption of no difference in habitat quality among different forest cover types, and vegetation composition and structure within a forest patch. The assumption that the selected forest bird species can use all the forest cover types is clearly an over-simplification. This leads to an over-estimation the area of forest that can be used by the selected bird species; the true area that can be used for breeding, for example, is likely to be smaller than the tree covered area because some species primarily use deciduous forests, for example. Conversely, habitat may be underestimated for more generalist species.

Another assumption concerns the habitat quality of the aggregated or generalized forest cover category. The tree communities of deciduous forests of the eastern U.S. change in composition with elevation, topography, and soil characteristics (Whittaker 1956). The assumption is made to treat all these variations as if they did not exist; one uniform classification of forest cover is used and it is assumed to have the same habitat value (quality) for the selected forest breeding birds. This generalization is necessary to focus the analysis on the broad pattern of the forest bird-habitat relationship and can be justified at the continental-scale analysis such as this study.

The use of land cover map and the subsequent aggregation of different forest cover types into one forest cover class preclude me from distinguishing among different tree species (e.g., deciduous or evergreen), height, vertical vegetation structure and composition within a patch, successional stages, or quality of tree covers (and thus, no habitat quality distinction can be made). For the land cover map, distinction among different forest cover types (i.e., coniferous, deciduous, mixed forest cover) is eliminated and treated as equal. However, the strength of forest vs. non-forest (binary representation) and/or one forest cover class is that (1) the findings can be generalized to broad urban regions in the eastern U.S. to extract general patterns of relationships between the amount and spatial configuration of forest cover and the distribution and abundance of forest bird species and (2) I can focus on the forest cover class, its composition and spatial configuration in a landscape.

1.4.3 Inherent Biases in the BBS Data

The North American Breeding Bird Survey (BBS) is suited for this kind of broad-scale study because it monitors populations of birds over a wide geographic area. The BBS provides long-term species-wide census data (Morrison 1986). The BBS “may be used to note changes (gradual or sudden) in populations,” enabling population trend analysis (Morrison 1986). BBS routes are established using a random stratified design so that they would be a good representation of the ecosystems of each state (Sauer et al.

2003). Nonetheless, the spatial distribution and density of routes varies, with the highest density occurring in the northeastern U.S. (Peterjohn et al., 1995), and spatial autocorrelation has been documented in analyses of BBS data (Flather and Sauer 1996, Thogmartin et al. 2004). It is also a concern that in the areas where the routes are less dense, they may not be picking up the signature of bird population change as compared to the areas where the route density is higher.

Moreover, because the BBS routes follow secondary roads, the land cover types found next to the roads may potentially be biased towards fragmentation. For example, the land use/cover around the roads may be more disturbed than that of the interior of intact forest patches. Also, since secondary roads by definition do not go through densely-built areas, the roadside land cover types do not include dense urban areas. In sum, the location of the BBS routes may bias the types of land cover included in the areas at certain distances from the routes. Betts et al.'s (2007b) study suggests that roadside vegetation may change at a different rate than that in a surrounding larger landscape, which can be a problem in regions characterized by rapid habitat alteration.

Other inherent biases in the BBS data include observer effects, issues of detection probability, and species geographic (natural) ranges. The observer effects and the natural ranges both affect the detection probability of certain bird species. The difference in detection probability among species is a problem because "the number of species observed in an area is determined jointly by the number of species actually present and by their respective probabilities of being detected by the observer" (Boulinier et al. 2001). Observer effects are the difference in observers' abilities to detect birds (Peterjohn et al. 1995). The difference can also stem from the changes of observers in charge of specific routes (Peterjohn et al. 1995) as well as more subtle effects of the same observer's skill level increase from the first year to subsequent years (Kendall et al. 1996). Observer difference may cover up actual increasing/decreasing trend (Veech 2006). At the edge of species geographic ranges, the number of the individuals of the species naturally declines (Veech 2006). This affects the probability of the species' detection. Careful interpretation of the result is required for the cause of this decline because bird abundance can be low at the edge of the species natural range, or due to land use/cover change, and other factors.

It is definitely a limitation in the analysis that the length of a survey route is much longer than the buffer distances. Even the widest buffer (6 km) is one-seventh of the total length (40 km). Therefore, any connectivity measure must be carefully interpreted given this limitation. Or, connectivity measures not affected by the truncation effect should be used. By the same token, only total counts of the species on the entire route are available, not at each stop (observation station). Because the route is long, the surrounding landscape structure from one end of the route can be quite different from that of the other end. The analysis must take this into consideration when analyzing the data.

Because the BBS is a roadside survey, the route itself is always classified as the "urban" land cover type, which always dissects the total area in the middle because the buffer is created on both sides of the route. This leads to the inflation of the percent urban classification and the reduction in any connectivity measures. The effect is especially strong for small buffers such as 180 m (six cells) on both sides of a route.

These limitations are acknowledged and need to be reflected in the analysis of the data. Even with the limitations and inherent biases in the BBS data, it remains a valuable source of information for the status of North American birds at a continental scale and is thus useful for a broad-scale study such as mine.

1.5 Conclusion and the Overview of the Chapters

Biodiversity loss is a global concern and land use decisions through habitat loss, fragmentation, and degradation greatly affect it. Because planning and design, by definition, change spatial configurations and consequently affect ecological processes, planners and landscape architects should be aware of the consequences of their actions through understanding of biodiversity and its functions (Ahern et al. 2006). Using forest birds as focal species, the study investigates route-level forest bird abundance with regard to forest loss and fragmentation in urban regions across eastern U.S. Urban regions are where most people live in the U.S. (Hobbs et al. 2002) and often coincide with the areas of high biodiversity conservation priority (Groves et al. 2000, Balmford et al. 2001, Araújo 2003). An urban region is also a relevant scale of planning/design/management, especially for species such as predatory species and neotropical migrants that have a large home range and a long dispersal distance. Therefore, if we want to protect biodiversity, it needs to be explicitly integrated into land-use plans for urban regions (Ahern et al. 2006). The study’s planning implications are then used in the development of a greenspace conservation planning framework for urban regions, incorporating important landscape planning and ecological concepts.

The dissertation chapters that follow are organized as follows. Chapter 2 reviews some important landscape ecology theories and concepts for their integration into landscape ecological planning. The chapter also traces the evolution of ecological planning, mostly in the U.S. with some European influences. It also discusses some important spatial and landscape ecological planning strategies and concepts. Chapter 3 presents the study of route-level, multi-scale analysis of forest bird abundance-habitat relationships in urban regions across the eastern U.S. Chapter 4 develops a landscape planning meta-model, building on resilience thinking and synthesizing previous general landscape planning models in the landscape planning best practice model, which becomes part of the meta-model. The chapter also demonstrates the conceptual application of the meta-model to greenspace conservation planning for the purpose of conserving forest bird populations at the urban regional scale, drawing on the results of the comparative observational study in the preceding chapter. Chapter 5 presents my overall conclusions based on the forest bird-habitat study (Chapter 3) and the development and conceptual application of the landscape planning framework (Chapter 4). The final chapter also discusses the implications of these results for planning, design, and management.

CHAPTER 2

REVIEW OF THE LITERATURE

This chapter describes literature relevant to the research hypothesis and questions of this dissertation. It is organized into four sections: (1) threshold concept and its potential application to landscape planning, (2) the use of the term and effects of habitat fragmentation, (3) connectivity from a landscape ecological perspective, and (4) application of landscape ecological theories and principles to landscape planning. In each section, the relevance of the literature to the research reported in this dissertation is discussed.

2.1 The Concept of Threshold and Its Potential Application to Landscape Planning

2.1.1 Introduction

The concept of threshold has a potential in application to landscape planning, especially conservation planning of species, habitats, and ecosystems. It also has significance in the management of social-ecological systems from a resilience perspective. However, our understanding and the use of threshold has been scattered among various disciplines, and the link to conservation planning and social-ecological system management for resilience has not been established very strongly. To fill in the gap, first, a review is conducted on how the term threshold is defined in dictionary and used in natural sciences and other fields where searching for thresholds is a common research topic by quick sampling of some representative fields. Threshold in the context of resilience is also reviewed as it arguably has significance for the management of social-ecological systems. Second, the application of threshold concept to watershed planning, as an example of landscape planning, is discussed. The advantage and challenges of a threshold approach is discussed. Finally, the link between threshold to conservation planning and policies, and social-ecological system management for resilience is firmly established.

2.1.2 Definitions, Use, and Characteristics of Threshold

2.1.2.1 Dictionary Definitions and Use of Threshold in Natural Sciences

According to the Merriam-Webster's online dictionary, the definitions of "threshold" that are relevant to this research include: (1) "a level, point, or value above which something is true or will take place and below which it is not or will not" and (2) "the point at which a physiological or psychological effect begins to be produced" (Merriam-Webster Online Dictionary 2009). The first type of threshold triggers a yes-or-no type of binary response and the second definition infers a point at which some effects begin to set in. What is common in both definitions is that a threshold is a point above which something takes effect and below which it does not. Oxford Dictionary of English offers the same definition of threshold: "the magnitude or intensity that must be exceeded for a certain reaction, phenomenon, result, or condition to occur or be manifested" (Soanes and Stevenson 2003). Under this core definition, threshold is used to mean "the maximum level of radiation or a concentration of a substance considered to be acceptable or safe" or "the level at which one starts to feel or react to something" (Soanes and Stevenson 2003). These definitions indicate a certain minimum level above which some effect takes place.

In natural sciences, a threshold is a point or zone of the value of an independent parameter where a small, additional change in the independent parameter causes sudden, large changes in the state of the dependent parameter. The large, sudden change of the dependent parameter (or the state it is in) is the characteristic of a threshold response. When the relationship between a dependent variable and an independent variable is plotted, a threshold is apparent by the discontinuity of smooth, gradual changes of the dependent variable when the threshold is crossed (Figure 2.1). For example, the survival probability of simulated populations suddenly drops to near zero when the available percentage of habitat in a landscape is reduced below a certain level (Fahrig 2001). Non-linearity of the relationship between independent and dependent variables is characteristic to the threshold response (Muradian 2001, Wiens et al. 2002, Cowling and Shin 2006, Serra et al. 2006).

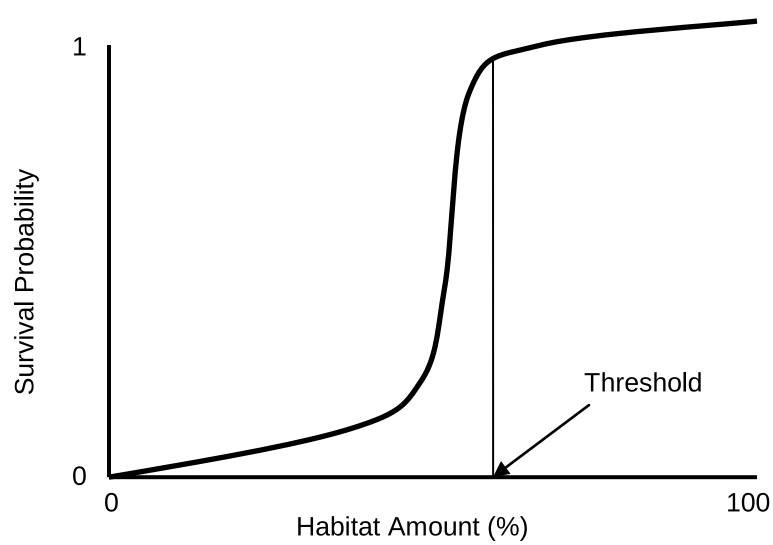

Figure 2.1: Extinction threshold. When remaining habitat is reduced below the threshold, the probability of population survival suddenly drops to near zero. A small additional loss of habitat near the threshold causes the sudden, large changes in the survival probability (redrawn from Figure 1 in Fahrig 2001).

2.1.2.2 Use and Definition of Threshold in Other Disciplines

In other fields where searching for thresholds is a common research topic, such as psychology, medical research, and public health, the term threshold is used in a similar manner. For example, in psychology, Garcia et al. (2007) conducted an experiment on gender differences in the pressure pain threshold: a level of pressure above which a participant feels pain. In Sokolov and Pavlova's (2006) experiment on visual motion detection, the term threshold was used to refer to the minimum displacement distance. These uses are consistent with the other uses and a dictionary definition that a threshold is some minimum point or value at which some effects set in. In medical research, Davis (1985) reviewed the concept of an anaerobic threshold: an exercise-induced lactic acidosis occurring at a particular oxygen uptake. In public health research, Georgette (2007) developed a model to calculate the fraction of population needed to be immunized during infectious disease outbreak to achieve herd immunity. The author called the fraction of population the herd immunity threshold: the minimum percentage of people that needs to be immunized to prevent disease outbreak. Bell et al. (2006) evaluated whether a "safe" threshold tropospheric ozone concentration level exists below which risk of premature mortality is not a human health concern. Both studies used the term threshold to mean some minimum value to guarantee some effect to take place or not to take place. In conclusion, in other fields where threshold research is common, such as psychology, medicine, and public health, the use of the term threshold is consistent with the dictionary definitions and the use in natural sciences: a threshold is a point, level, or value above or below which the state of a response variable drastically changes.

2.1.2.3 Threshold in the Context of Resilience

Resilience is defined as "the capacity of a system to absorb disturbance; to undergo change and still retain essentially the same function, structure, and feedbacks" (Walker and Salt 2006). In other words, resilience is the capacity of a system to absorb disturbance without shifting to another regime (Holling 1973, Walker et al. 2004, Walker and Salt 2006). The shift to another regime occurs when a threshold is crossed as thresholds exist between alternative regimes in social-ecological systems such as ecosystems and landscapes (Folke et al. 2004, Walker and Salt 2006). Here, "regime" and "stable state" both mean a set of states within which a system tends to stay (Walker and Salt 2006).

The existence of alternative regimes or multiple stable states has documented in various ecological, social, and social-ecological systems around the world (Carpenter 2001, Gunderson and Pritchard 2002, Holling and Gunderson 2002, Folke et al. 2004, Walker and Meyers 2004, Resilience Alliance and Santa Fe Institute 2009). The examples of regime shifts include changes in vegetation from sawgrass (*Cladium jamaicense* Crantz) to cattails (*Typha domingensis* Pers.) in the Everglades, Florida, U.S.A. (Gunderson and Pritchard 2002), changes from grass-dominated savanna to shrub-dominated savanna (Scheffer et al. 2001, Bestelmeyer et al. 2003), eutrophication of a lake system (Scheffer et al. 2001, Carpenter 2003), collapse of fisheries (Folke et al. 2004, Walker et al. 2004, Walker and Salt 2006), salinization of an agricultural basin in Australia (Folke et al. 2004, Walker et al. 2004, Walker and Salt 2006), and decline of corals and increase of brown algae in the Caribbean area (Nyström et al. 2000). These are all examples of a system crossing a threshold and flipping into a different (often not desirable) stable state (Walker and Salt 2006).

Thresholds can be visualized in the conceptual framework that depicts a social-ecological system as a ball in a basin (Folke et al. 2004, Walker et al. 2004, Walker and Salt 2006, see Figure 2.2). The “state space” of a system is defined by the (state) variables that constitute the system (Walker et al. 2004). For example, a suburban neighborhood can be defined by the median income of household, the mode of transportation, and the ethnic composition of the community. A “basin of attraction” is a region in state space in which the system tends to remain (Walker et al. 2004). Each basin represents a set of states with the same kinds of functions and feedbacks. There may be more than one basin of attraction for any given system—alternative stable states (i.e., alternative regimes) (Walker et al. 2004, Walker and Salt 2006). The various basins that a system may occupy and the boundaries that separate them consist of a “stability landscape” (Walker et al. 2004). Thresholds are the edges of basins. In the metaphor of a ball in a basin, the position of the ball in the stability landscape represents the current state of the system (Walker et al. 2004, Walker and Salt 2006). Although the ball is attracted to the bottom of a basin, representative of an equilibrium state, because social-ecological systems are constantly affected by disturbances, stochasticity, and decisions of actors, the position of the ball keeps changing and the ball never stays at the bottom of a basin (Scheffer et al. 2001, Walker et al. 2004). Moreover, the stability landscape itself keeps changing due to external drivers (e.g., temperature, grazing pressure) and internal processes (e.g., nutrient cycling, predator-prey cycles, management practices), leading to changes in the number of basins, in the positions of the basins within the state space, in the position of thresholds (edges) between basins, or in the depths of basins (Walker et al. 2004). Due to the external forces and internal processes of a system, and changes in the stability landscape, when the system crosses some limit (the edge of the basin), the feedbacks that drive the system's dynamics change, and the system moves toward a different equilibrium. The potential exists for sudden transitions of systems from one basin of attraction to another, which fundamentally change the qualitative nature of systems (van de Koppel et al. 1997, Scheffer et al. 2001, Folke et al. 2004, Walker et al. 2004).

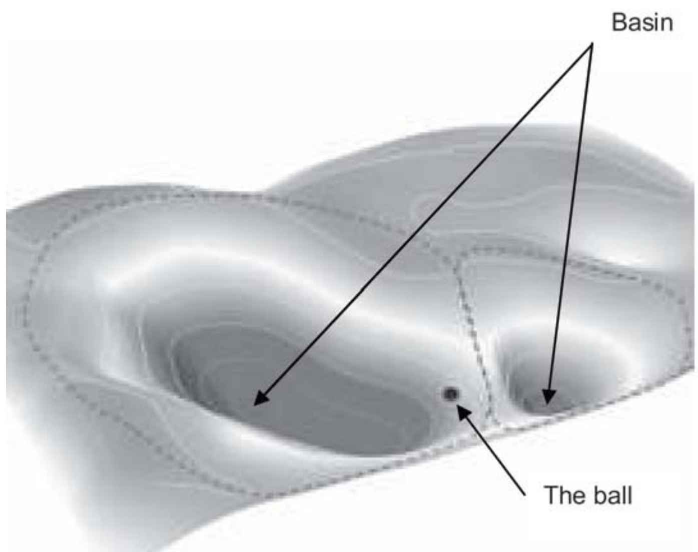

Figure 2.2: The metaphor of a system as a ball in a basin. The ball is the current state of a social-ecological system. A "basin of attraction" is a region where the system tends to remain; each basin represents a set of states with the same kinds of functions and feedbacks. The dashed line is a threshold separating alternative regimes (modified from Figure 1b in Walker et al. 2004, who acknowledge Art Langston for the construction of the figure; used with permission from Brian Walker).

From a resilience perspective (Holling 1973, Gunderson and Holling 2002), the question is how much change can occur in the basin and in the system's trajectory without the system leaving the basin (Walker and Salt 2006). Because crossing a threshold leads to qualitative changes to the system (including often undesirable states), managing the amount of disturbance the system receives and knowing how much disturbance it can take is critical for the system's resilience (Walker et al. 2004, Walker and Salt 2006). To mange and enhance the resilience of a social-ecological system, it is crucial to (1) identify the drivers (i.e., slow, controlling, coarse-scale variables coupled with fine-scale, fast variables) that cause a social-ecological system to cross thresholds between alternative stable states, (2) identify the thresholds on the drivers, and (3) enhance aspects of the system that enable it to maintain its resilience (Walker and Salt 2006). Detection of thresholds, however, is not straightforward and requires cautious analysis (Carpenter 2001).

In conclusion, threshold in the context of resilience is significant for the management of social-ecological systems because crossing thresholds means that a system is entering a qualitative different state with a different set of dynamics and feedbacks and this state is often undesirable in terms of sustainable production of ecosystem services (e.g., reduced biodiversity, polluted water, etc.). Therefore, for the management of social-ecological systems for resilience, it is important to identify on what variables thresholds exist and when a system may cross the thresholds to flip to an undesirable state. Building the capacity to manage the system in relation to these thresholds leads to achieving sustainability (Walker et al. 2004, Walker and Salt 2006).

2.1.2.4 Point-type and Zone-type Thresholds

Thresholds can be points or zones. The examples of point thresholds include: the physics of phase transitions, the potential effect of global warming on the Gulf Stream, and the change of transparency of a glass of Pernod by addition of water. The first example of point thresholds is physical state changes among ice, water, and vapor by temperature changes. The physical state of water changes from solid to liquid at precisely 0°C and from liquid to vapor at precisely 100°C. The second example is the potential changes in the Gulf Stream due to global warming. Bunyard (2004) warns that if global warming continues, at the critical unknown temperature, the Gulf Stream will either halt or shift much further south, resulting in a temperature decrease in northern Europe. What Bunyard is concerned with is not only that this will likely happen but also the suddenness at which the flowing Gulf Stream can stall and the temperature can drop over northern Europe: it can all happen in a matter of years—which is extremely fast in geologic or climatic time—not centuries or millennia. The third example of point-type thresholds is familiar to the drinkers of Pernod. When a small amount of water is added to a glass of Pernod, initially, the transparency does not change. However, in the process of adding the water drop by drop, there comes a moment at which the mixture becomes opaque. In all threshold changes, "at a phase transition, a system changes its behavior qualitatively for one particular value of a continuously varying parameter" (Stauffer and Aharony 1994).

Although most thresholds are a point on a continuous independent variable, some thresholds occur as a zone on a continuum. The difference between point-type and zone-type thresholds is that the latter involves a more gradual (but still non-linear) transition between states rather than an abrupt change, which is the characteristic response of point-type thresholds (Muradian 2001). Muradian (2001) proposed a "threshold zone": a zone of transition between states. For example, Muradian (2001) suggests the existence of such a threshold zone for the relationship between equilibrium island species numbers and island size based on the study of Ward and Thornton (1998) (Figure 2.3). A shift to a different trajectory of island species richness would occur somewhere in the zone of island sizes.

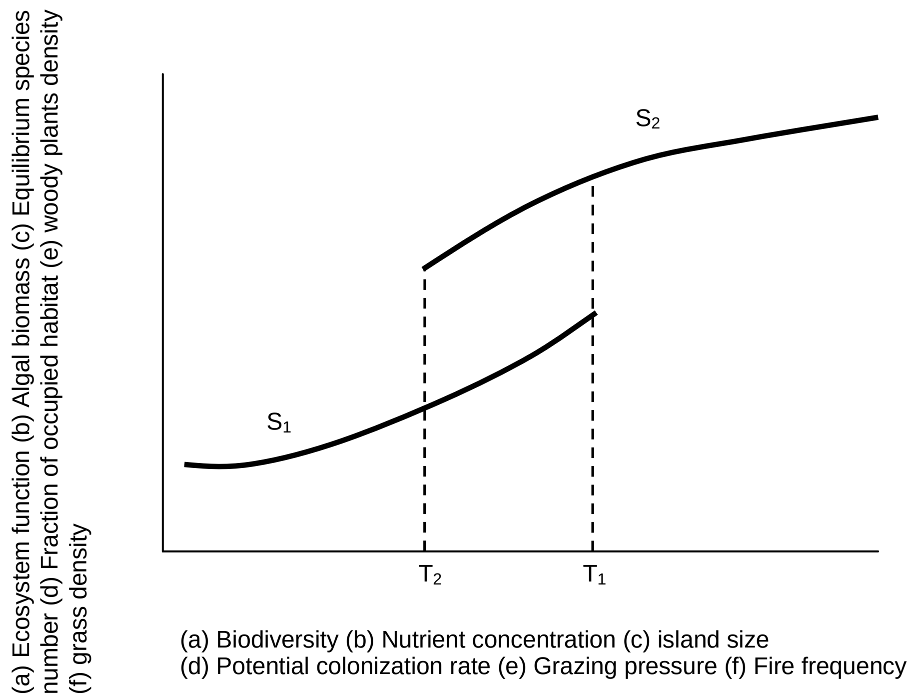

Figure 2.3: Hypothetical relationship (but based on empirical data) between two parameters, showing a zone-type threshold. (a) through (f) are examples. The example of (a) on the X axis corresponds with the state change of (a) on the Y axis, and so on. Sudden shifts to alternative states occur somewhere within a zone (between T_1 and T_2) (redrawn from Figure 1 in Muradian 2001).

Wiens et al. (2002) considered thresholds as regions or zones, in which the points within the zone of transition may have similar rates of change than those outside the zone (Wiens 1989, Case 2000). This would suggest that if key species and/or ecological processes are monitored closely, the changes in the rates may be detected, enabling intervention before irreversible change occurs (Wiens et al. 2002). This highlights the importance and potential of proactive planning based on thresholds. If we could reasonably assume that independent variables (structural changes) can act as surrogates for dependent variables (e.g., ecological processes), it would make the life of natural resource managers and planners much easier because structural variables are usually easier to quantify and monitor (e.g., by using landscape metrics) than ecological processes (Havstad and Herrick 2003). In the context of the conservation of forest species, if there is a threshold of forest amount below which the number of individuals suddenly declines, the threshold may aid in taking a proactive conservation action before the amount of forest in a region is reduced below that level or serve as a useful target of restoring the amount of forest to that level. This would translate to conservation planning actions such as prioritizing land management for protection or acquisition options, and targeting areas for restoration.

2.1.3 Application of the Threshold Concept to Watershed Planning

One way in which thresholds are used to guide or control land development is based on the studies that demonstrated the existence of threshold percentage of impervious surfaces in a watershed for stream degradation both physically and biologically (i.e., stream quality) (Schueler 1994, Arnold and Gibbons 1996). Moglen and Kim (2007) define impervious surfaces as “human-produced surfaces that are essentially impenetrable by rainfall.” There have been studies that linked the increase in impervious surfaces to the decrease in stream quality, or water-related environmental degradation in general (Schueler 1994, Arnold and Gibbons 1996, Booth and Jackson 1997, Wang et al. 2001, Booth et al. 2002, 2004, Brabec et al. 2002, Center for Watershed Protection 2003). Increased impervious surfaces degrade streams both physically and biologically by higher volumes of surface runoff entering streams, faster arrival in streams, and poor quality of runoff (Center for Watershed Protection 2003, Moglen and Kim 2007). The studies that investigated the relationship between impervious cover and various measures of stream degradation (e.g., various biotic indices such as fish and benthic invertebrate community composition and diversity) indicate a threshold of 10-15% of impervious surfaces in a watershed beyond which stream quality starts to decline rapidly (Schueler 1994, Booth and Jackson 1997, May et al. 1997, Wang et al. 2001, Miltner et al. 2004). For example, Wang et al. (2001) identified a threshold region between 8 and 12 % of connected imperviousness (i.e., effective impervious area) above which a small increase in the percent imperviousness in a watershed leads to rapid changes of stream quality—characteristic of critical threshold phenomena. Booth et al. (2002), however, show a continuum of biological response to a level of imperviousness, not a threshold response—especially, a wide range of responses to a low percentage of impervious surfaces.

Since urbanization increases imperviousness incrementally by adding roads, rooftops, parking lots, side walks, and other impervious surfaces, the existence of the threshold is then used to "justify limiting imperviousness to protect stream conditions" (Moglen and Kim 2007) in many communities that face the problem of water quality degradation of their streams, lakes, and bays due to urbanization (e.g., Center for Watershed Protection and Maryland Department of the Environment 2000, King County, Washington 2009, Miami-Dade County, Florida 2010). To protect stream conditions, the results of the studies that indicated the existence of threshold have lead to policy recommendations "to limit the amount of imperviousness in new development to values less than an identified threshold" (Moglen and Kim 2007). The U.S. Environmental

Protection Agency (U.S. EPA), for example, suggests the 10% threshold as a general guideline for watershed-based zoning plans (Kwon et al. 2008).

Moglen and Kim (2007), however, caution the use of a fixed threshold value such as 10% for watershed planning because: (1) difference in measuring imperviousness (e.g., land cover based or land use based) can result in large differences in percent imperviousness values (Schueler 1994, Brabec et al. 2002); (2) various metrics of stream quality used to document the impervious surface-stream quality relationship are not always comparable with each other (Schueler 1994, Brabec et al. 2002, Booth et al. 2004); and (3) the spatial distribution of imperviousness has a large effect on aggregate imperviousness and consequently, the water quality of the watershed (Brabec et al. 2002); even if the point measurements at the outlets of major watersheds are below the threshold, all locations along the stream network may not. Also, stormwater best management practices (BMPs) implemented at site and neighborhood scales can cumulatively exert a positive influence on removing pollutants and reducing first-flush events (Strecker et al. 2001); their positive effects may not be detected by the percent imperviousness. The caveat in using a standard threshold value for watershed planning is similar to the issues of using only one threshold percent habitat value for species conservation. Using only one threshold value can lead to over-simplification since the threshold habitat amount varies among species, affected by species' life history traits, habitat configuration, and the matrix quality (Fahrig 2001, Radford and Bennett 2004).

Therefore, instead of relying only on the regulation of impervious surfaces for new developments and retrofitting projects, Booth et al. (2002) recommend a more integrated solution to protect aquatic resources from development and to mitigate development impacts. Their recommendations include: impervious-surface limits, forest-retention policies, stormwater detention, riparian-buffer maintenance, and protection of wetlands and unstable slopes. Relying on one index such as the percentage of threshold in a watershed can be a cost-effective method but a more holistic approach that integrates land uses in a watershed and their spatial configurations, and also models that link environmental and economic considerations (e.g., Randhir and Shriver 2009) are necessary to reduce environmental impact of future development.

In this section, I have used watershed planning as an example to demonstrate the linkage between the concept of threshold and its potential application to landscape planning. Threshold-based watershed planning is based on the demonstrated threshold effect of the percentage of impervious surfaces in a watershed on stream biological and physical quality. The percentage of impervious surfaces in a watershed can be used to guide or control land development to protect stream quality.

2.1.4 Conclusions

Threshold can be a point or a region; either way, if it is crossed, some effect takes place. Moreover, a small, additional change in an independent variable triggers sudden, large changes in the state of a dependent variable as the threshold is crossed. For example, simulation studies have shown an extinction threshold, which is the amount of habitat in a landscape below which the probability of population survival suddenly drops to near zero. Non-linear relationship characterizes the threshold response. In the context of resilience, threshold is related to regime shift. When a threshold is crossed, a social-ecological system flips to an alternative stable state. Because an alternative stable state may not necessarily be a desirable one, the task for natural resource managers and planners is to increase the resilience of the system so that it remains in a desirable regime, or actively navigate away from a current undesirable regime. To mange a social-ecological system for resilience, it is crucial to (1) identify slow, controlling variables that cause a system to cross thresholds between alternative stable states, (2) identify the thresholds on the drivers, and (3) enhance aspects of the system that enable it to maintain its resilience (Walker and Salt 2006).

The application of the concept of threshold to watershed planning is based on empirical studies that have shown the existence of threshold percentage of impervious surfaces in a watershed for both physical and biological stream degradation. Various measures of stream quality such as fish and benthic invertebrate community composition and diversity show a sign of rapid degradation when the percentage of impervious surfaces in a watershed exceeds 10-15% (Schueler 1994, Booth and Jackson 1997, May et al. 1997, Wang et al. 2001, Miltner et al. 2004). Since urbanization incrementally adds impervious surfaces, the existence of the threshold is used as a scientific basis to justify a policy that regulates the amount of imperviousness in a new development (and retrofitting old ones) to protect stream conditions in many communities that face water pollution and water quality degradation. However, there is a danger in the “one-threshold-fits-all” type of approach due to the difference in the measures of impervious surfaces and stream quality (Schueler 1994, Brabec et al. 2002, Booth et al. 2004). Moreover, the spatial distribution of impervious surfaces within a watershed affects water quality even with the same percentage of impervious surfaces (Brabec et al. 2002) and the positive effects of stormwater BMPs may not be detected by the percentage of impervious surfaces alone (Strecker et al. 2001). Therefore, a more holistic approach to stream quality degradation is recommended, including the consideration of the location of land uses within a watershed, the application of various stormwater BMPs, and the protection of riparian vegetation, wetlands, and steep slopes in addition to impervious-surface limits.

There is a potential for the application of threshold to landscape planning, especially the conservation of species, habitat, and ecosystems. If a threshold can be identified for a slow, controlling, structural variable, as suggested by the resilience concept, the use of threshold can become an attractive, cost-effective method to engage in proactive planning. For example, in the context of the conservation of forest species, if there is a threshold of percent forest cover (an independent variable) below which the number of individuals of forest birds and mammals (a dependent variable) suddenly declines, the threshold may aid in taking a proactive conservation action before the forest cover in a region is reduced below this level or serve as a useful target of restoring the forest cover to the threshold level. This would translate to the development of conservation planning policies that would prioritize land management for protection or acquisition options, and target areas for restoration. The difficulty in establishing such thresholds is the lack of species-specific data on the factors that affect the thresholds such as life history traits, movement ability, and the permeability of the landscape matrix.

Watershed planning is another example of the potential application of the threshold concept to landscape planning of watersheds. A critical variable, the percentage of impervious surfaces in a watershed (an independent variable), has been demonstrated to affect stream quality (a dependent variable), and it has a threshold. The existence of the threshold is used as a scientific basis to develop land-use planning policies to limit the amount of impervious surfaces in new developments and even to retrofit existing ones to protect stream conditions.

Although there is a growing database of demonstrated and proposed thresholds in ecological, social, and social-ecological systems (Resilience Alliance and Santa Fe Institute 2009), because of the complexity of real systems and the difficulty of identifying thresholds, the cases for known thresholds are still sparse for various systems. In social-ecological systems, thresholds correspond to the boundaries between alternative stable states. When they are crossed, qualitative changes occur to the system state and there is a possibility that the system flips to an undesirable state from the human well-being perspective, such as to an alternative state with reduced and degraded ecosystem services. Threshold-based landscape planning and management of social-ecological systems for increased resilience capacity can enable more proactive planning based on identified thresholds. Important indicators should be monitored and identified thresholds can be used as policy and planning targets so that the variables with thresholds, such as forest cover and impervious surface, can be protected and/or restored to achieve the targets. Although due caution needs to be exercised not to rely solely on the identified thresholds to achieve desired planning and management goals, there is a potential in the threshold-based approach to landscape planning and the management of a social-ecological system for resilience, and the research efforts should continue to identify key system drivers and the thresholds on them so that undesirable stable states can be avoided before the threshold is crossed.

2.2 The Use of the Term and Effects of Habitat Fragmentation

2.2.1 Common Grounds

2.2.1.1 Introduction

Habitat fragmentation refers to the breaking apart of contiguous habitat into smaller pieces (Forman 1995, Fahrig 1998, 2003, McGarigal and McComb 1995, D'Eon 2002). The terms patch and fragmentation are discussed here in terms of wildlife habitat value (usually for particular species in mind, e.g., birds). The patch and fragmentation terms can be applied to other perspectives or values such as recreation, transportation, hydrology, and agriculture. For example, we can examine the effect of fragmentation of agricultural lands on the effectiveness of grain production in urbanizing counties in Iowa. Although these other values are equally important in developing landscape plans that can serve multiple purposes, I will focus on patches as wildlife habitat.

The term fragmentation is discussed and dealt in the dissertation in the framework of a patch-corridor-matrix model (*sensu* Forman and Godron 1981, Forman 1995) of representing a landscape and its composing spatial elements (e.g., forests, fields, water bodies, and developed areas). The patch-corridor-matrix model is based on the assumption that horizontal landscape elements can be distinguished by clear boundaries (Forman 1995). Although vertical landscape attributes are as important, in landscape ecological studies, the research has focused on the relationships among horizontal landscape elements (e.g., land uses and ecosystems) and their effects on ecological processes (Risser et al. 1984, Turner 1989, 2005, Zonneveld 1990, 1994, Pickett and Cadenasso 1995, Turner et al. 2001, Wu and Hobbs 2007b).

2.2.1.2 Habitat, Patch, and a Patch-Corridor-Matrix Model

Habitat refers to "the place where an animal or plant normally lives, often characterized by a dominant plant form or physical characteristic (that is, the stream habitat, the forest habitat)" (Ricklefs and Miller 2000, p. 731). Habitat therefore includes the necessary resources and conditions for specific organisms for their specific purposes such as foraging and nesting (Ricklefs and Miller 2000).

Patch is a fairly homogenous and nonlinear area that is distinct from the surrounding landscape (Forman and Godron 1986, Forman 1995). Habitat patch, therefore, can be defined as a relatively homogeneous and distinguishable area (unit) in a landscape that supports the specific need and activity of the organism/species/population during a specific life stage. I use the term patch interchangeably with habitat patch for a specific species. For example, habitat patch for Wood Thrush (*Hylocichla mustelina*) during its breeding season is the interior and edges of deciduous and mixed forests (Roth et al. 1996, Gough 2007). In this research, patch always refers to a forested patch and forest is the habitat for the selected woodland breeding bird species. Earlier I discussed this assumption that "forest" land cover is the habitat for the selected forest bird species (see section 1.4.2).

Patch is a term defined in the patch-corridor-matrix model of a landscape (Forman and Godron 1981, Forman 1995). The model represents a landscape as a mosaic of categorical, heterogeneous landscape elements with discrete boundaries, and these landscape elements are classified into patches, corridors, and the matrix (Forman and Godron 1981, 1986, Forman 1995). The model is best applicable to a landscape where distinct boundaries between different land covers/uses can be recognized; for example, an agricultural landscape where agricultural fields are interspersed with hedgerows and occasional remnant forests. On the contrary, the patch-corridor-matrix model does not work well when the boundaries are fuzzy and habitat patches are difficult to distinguish such as in forest mosaics (Betts et al. 2006b).

The patch-corridor-matrix model can be applied to both an island biogeographic perspective and a landscape mosaic perspective. The island biogeographic perspective treats habitat patches as the “islands” in the inhospitable “sea” of unsuitable habitat. The landscape mosaic perspective is taken when full spatial heterogeneity is embraced: a landscape is composed of various landscape elements (e.g., land use/cover types) with each having varying habitat values to a particular organism. The simplicity of the binary classification of a landscape into habitat and non-habitat is the strength of the island biogeographic model. The weakness in the models based on the theory of island biogeography (MacArthur and Wilson 1967) and the metapopulation theory (Levins 1970) is that they disregard the habitat value of the matrix in which (habitat) patches are embedded (Haila 2002). Also, unlike the ocean, which is largely impassable, the land matrix may not present significant barriers to organisms moving through the landscape (D’Eon 2002). The models fail to accommodate for the way specific species perceive and use heterogeneous landscapes (Wiens et al. 1993, Ricketts 2001, Bender and Fahrig 2005). Moreover, the temporal evolutional forces acting on oceanic islands are different from those on habitat patches in land mosaic (Haila 2002). These weaknesses can be addressed by incorporating the landscape mosaic perspective—that a landscape is composed of patches of various types, not as simple binary classification of habitat and non-habitat but different patch types influencing ecological processes to varying degree.

For example, the landscape mosaic perspective can be incorporated by functional metrics such as edge contrast—although the metric requires researchers to assign weights to the degree of edge contrast for all pairwise combinations of different patch types—and resistant kernels to create cost surfaces for organism movement/dispersal (see Compton et al. [2007] for this application). The patch-based models (i.e., the models based on the island biogeographic and landscape mosaic perspective) are contrasted with a continuous or gradient representation of a landscape, which conceives the landscape with underlying, continuously varying abiotic parameters (environmental gradients) affecting the abundance and distribution of organisms (McGarigal and Cushman 2005).

Even though the patch-based models have certain limitations, they still serve as a useful framework to represent and study a landscape. Also, there are well-established tools (e.g., FRAGSTATS) and methodologies (e.g., analysis of variance) to work with the framework (Leitão et al. 2006). Therefore, I will use the models (based on the island biogeographic and landscape mosaic perspective) as a basic underlying framework of representing a landscape and will use the method and tools suitable to analyze the models.

2.2.2 Definition of Habitat Fragmentation

2.2.2.1 Habitat Fragmentation *Per Se*

Fragmentation is one of the five major types of spatial land transformation along with perforation, dissection, shrinkage, and attrition (Forman 1995). Fragmentation has a wide range of spatial, species, and other effects (Forman 1995). The most commonly accepted definition of habitat fragmentation is the breaking apart of contiguous habitat into smaller pieces (Forman 1995, Fahrig 1998, 2003, McGarigal and McComb 1995, D'Eon 2002). Habitat fragmentation is often accompanied with the loss of habitat (i.e., the shrinkage and/or complete removal of the broken-apart habitat) (Figure 2.4, Haila and Hanski 1984, Harrison and Fahrig 1995, Fahrig 1997, 1999, D'Eon 2002, Noss et al. 2006). Although the loss of total habitat area is a natural consequence of the subdivision of large, contiguous habitat, there are those among researchers who argue that the use of the term habitat fragmentation should be reserved for the breaking apart of habitat, independent of habitat amount (loss) (*sensu* Fahrig 1997, 2003). This narrow definition distinguishes habitat fragmentation *per se* from habitat loss. On the other hand, a broader definition of habitat fragmentation includes the loss of habitat in the former definition. In other words, habitat fragmentation is both the breaking apart of habitat and the shrinkage/loss of the remaining habitat (Wilcox and Murphy 1985, Noss 1991, Robinson et al. 1995, Schumaker 1996, Peck 1998, van den Berg et al. 2001, Hovel 2003).

Forman and Collinge (1996) conceive fragmentation as a phase in the broader sequence of land transformation. As noted above, the five spatial processes of landscape change are perforation, dissection, fragmentation, shrinkage, and attrition (see Figure 12.1 in Forman 1995). Typically, perforation and dissection are important in the beginning of land transformation, fragmentation in the middle of the sequence, followed by shrinkage and attrition as the percentage of original habitat type decreases from 100 to 0% (see Figure 12.2a in Forman 1995).

To summarize, aside from Forman's (1995) model of land transformation, habitat fragmentation in ecological literature has two definitions. The narrow definition focuses on the breaking apart of habitat into smaller pieces while controlling for changes in the amount of habitat (i.e., habitat fragmentation per se). The broad definition includes both habitat loss and fragmentation per se. As it will become evident, the argument for the more strict definition of habitat fragmentation cannot be separated from the discussion of habitat loss (Figure 2.4), which I will discuss next.

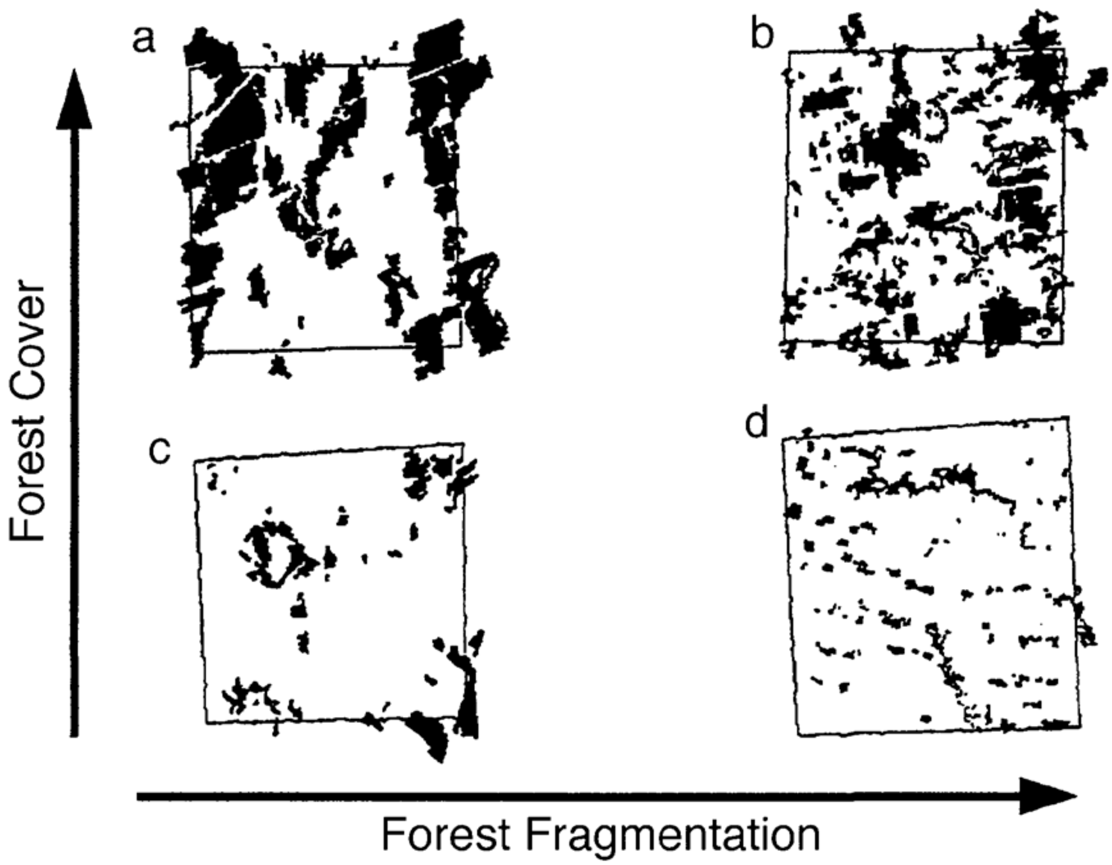

Figure 2.4: The black areas represent remaining forest habitats. The landscapes "b" and "d" are more fragmented than the landscapes "a" and "c." The landscapes "a" and "b" have more forest cover than the landscapes "c" and "d." The effects of forest fragmentation and forest loss are confounded in the conclusion that landscape "d" is more fragmented than landscape "a." Although this conclusion is correct, landscape "d" contains less forest. Therefore, if fragmentation per se is defined as the breaking apart of habitat, the effect of the reduced area of forest must be separated from that of fragmentation (defined broadly) of forest patches to truly measure the independent effect of forest fragmentation per se (*Source*: Figure 3 in Trzcinski et al. 1999).

2.2.2.2 Habitat Loss

The loss of habitat here refers to reduction in area of the particular habitat type of interest—for example, forest—not reduction in the number of habitat types (habitat diversity). Because oftentimes a researcher is focusing on a particular type of habitat for the species of interest, for example, forest habitat for forest dwelling mammals, s/he is usually not concerned with the loss of that habitat type itself (although it can certainly occur) but with the loss of (total) area that the particular habitat type occupies (or the percentage of that habitat type in a specified landscape) and the size of each habitat patch.

In this dissertation, I will mainly discuss the loss (i.e., reduction in area) of habitat patches not the loss of land cover types although this kind of loss inevitably occurs when a landscape loses habitat type diversity (e.g., converted to monoculture). Since my research focuses on forest cover type (forest as habitat for forest breeding birds), I will discuss changes in this particular habitat patch type. Therefore, in my dissertation, the loss of habitat applies only to forest habitat patches. Using FRAGSTATS, I plan to quantify forest cover (amount) as breeding habitat for the selected forest bird species, using the percentage of forest cover in a landscape. In studying habitat loss, it is important to know how much of the target patch type (habitat) exists within the landscape (McGarigal et al. 2002).

Habitat loss or destruction, along with habitat degradation and fragmentation, are the number one cause of global decline of biodiversity (Noss 1991, Tilman et al. 1994, Fahrig 1997, Peck 1998, Wilcove et al. 1998, Pullin 2002, Groom et al. 2006). When habitats disappear, together gone are the species that inhabit them. Once habitat is lost, it is difficult, costly, and takes time to restore it, and there is no guarantee that the species once inhabited it will return. Therefore, habitat loss (amount) has a large influence on the abundance and distribution of species; actually so large that its effect can mask the effect of habitat fragmentation per se (Simberloff 2000, Haila 2002). This necessitates the need to measure the relative effects of habitat loss and fragmentation per se to truly know how much each contributes to the abundance and distribution of species.

2.2.2.3 Contentious Issues

When the definition of habitat fragmentation includes habitat loss, it seems to be used more casually to equate low habitat proportions with high fragmentation due to increased distance between remaining habitat patches (e.g., Betts et al. 2007a). The proponents of the more strict definition of habitat fragmentation support the notion of fragmentation as only a spatial configuration phenomenon, independent of habitat loss (D'Eon 2002, Haila 2002). This is why in fragmentation studies some researchers call the spatial configuration of habitat as habitat fragmentation (e.g., Fahrig 2001, Cooper et al. 2002, Sleeman et al. 2005, Betts et al. 2006b) or isolation (e.g., D'Eon 2002) particularly to emphasize that habitat is "fragmented" or patchy as opposed to contiguous or clumped, often, without offering an explicit definition of fragmentation (e.g., With and King 2001, Wiegand et al. 2005). In some studies, when the spatial configuration of habitat (e.g., patch shape, patch isolation, proximity to edge) is characterized by various landscape pattern indices (or landscape metrics), habitat fragmentation is claimed to be quantified. Meyer et al. (1998) is a good example of claiming that they measured the effect of fragmentation by "fragmentation metrics"—these metrics are simply the measure of spatial configuration of habitat patches. The problem is that many of these landscape configuration metrics correlate with habitat amount (Trzcinski et al. 1999, Betts et al.

2006a). The use of these metrics by itself does not remove the effect of habitat loss. We would need appropriate experiment designs and/or statistical methods to remove the correlation (Fahrig 2003). The problem in all this is that many fragmentation studies do not clearly separate the effect of fragmentation per se from that of habitat loss (Schmiegelow and Mönkkönen 2002, Fahrig 2003). This needs to be done because confounding the effects of habitat loss and fragmentation could lead to erroneous conclusions and implications for conservation planning (more discussion on this later).

2.2.2.4 Isolation

2.2.2.4.1 Isolation and Habitat Fragmentation

Isolation is another concept deeply related to the issue of habitat fragmentation. In some studies, similar to how the term fragmentation is used, isolation is used as a general term for the spatial configuration of habitat (e.g., D'Eon 2002, Radford and Bennett 2004); in others, isolation is just one of many aspects of spatial configuration as fragmentation is (e.g., Hovel 2003). When researchers discuss isolation, they usually mean "patch" isolation: the degree to which neighboring habitat patches are apart. Patch isolation is measured as a distance from a focal patch (where various measurements such as bird count and vegetation composition are taken) to its neighboring habitat patches or to its nearest neighbor (i.e., the nearest neighbor distance) (e.g., Radford and Bennett 2004, Russell et al. 2005, Ferraz et al. 2007). In bird studies, isolation is measured as the distance to the nearest (occupied) patch (e.g., Fernández-Juricic 2004, Radford and Bennett 2004, Monteil et al. 2005) or as the percentage of habitat within a certain distance from a sample point/plot (e.g., Robbins et al. 1989).

In a landscape structure analysis software, FRAGSTATS, McGarigal et al. (2002) provide various metrics to measure spatial isolation. These are classified into two types: one that is based on Euclidean distance between nearest neighbors (McGarigal and Marks 1995) and the other on the cumulative area of neighboring habitat patches (weighted by nearest neighbor distance) within some ecological neighborhood (Gustafson and Parker 1992). These measures can be modified to take into account a landscape mosaic perspective. For example, simple Euclidean distance can be modified to account for functional differences among organisms. Isolation can also be measured by the degree of contrast (i.e., the magnitude of differences in one or more attributes between adjacent patch types) between the focal habitat and neighboring patches to account for the context of habitat patches (McGarigal et al. 2002).

Some researchers (e.g., Goodsell and Connell 2002, Russell et al. 2005) use the term habitat "proximity" interchangeably with isolation to mean distance between habitats. When fragmentation is used to mean the discontinuity of habitat in general, resulting in the decrease in connectivity, the representative aspect of fragmentation is isolation (proximity).

2.2.2.4.2 Effect of Isolation

One of the consequences of fragmentation is the increase in isolation (distance between patches) (D'Eon 2002, Noss et al. 2006). This leads to decrease in (landscape) connectivity, which consequently negatively affects movement/dispersal of organisms on a landscape (Tischendorf and Fahrig 2000a, b, Bender et al. 2003, Radford and Bennett 2004, Russell et al. 2005).

In their marine experiment, Russell et al. (2005) found that species mobility interacts with distance to neighboring habitats to create differing species assemblage composition. The composition of polychaetes, less mobile, benthic crawlers, differed between near and far habitats independent of habitat size; whereas, the composition of copepods, more mobile, water column swimmers, only differed between sizes of habitat when they were far apart. Ferraz et al. (2007) found a strong effect of area and a variable effect of isolation on the predicted patch occupancy by birds.

Since habitat loss alone at a landscape scale can lead to a reduction in number of habitat patches and inevitably lead to increased distances between habitats (Case B in Figure 2.5), the number of patches per unit area and the proximity of patches are not independent within each landscape (Goodsell and Connell 2002). If increase in distance between habitats (Cases A and B in Figure 2.5) is taken as increase in fragmentation of habitat (although this is not necessarily true if the narrow definition of fragmentation is applied), it appears as if habitat loss alone at a landscape scale could cause habitat fragmentation. This can lead to a confusion of the effect of habitat loss with that of habitat fragmentation, and this is why fragmentation needs to be studied at a landscape scale (detail discussion on this later).

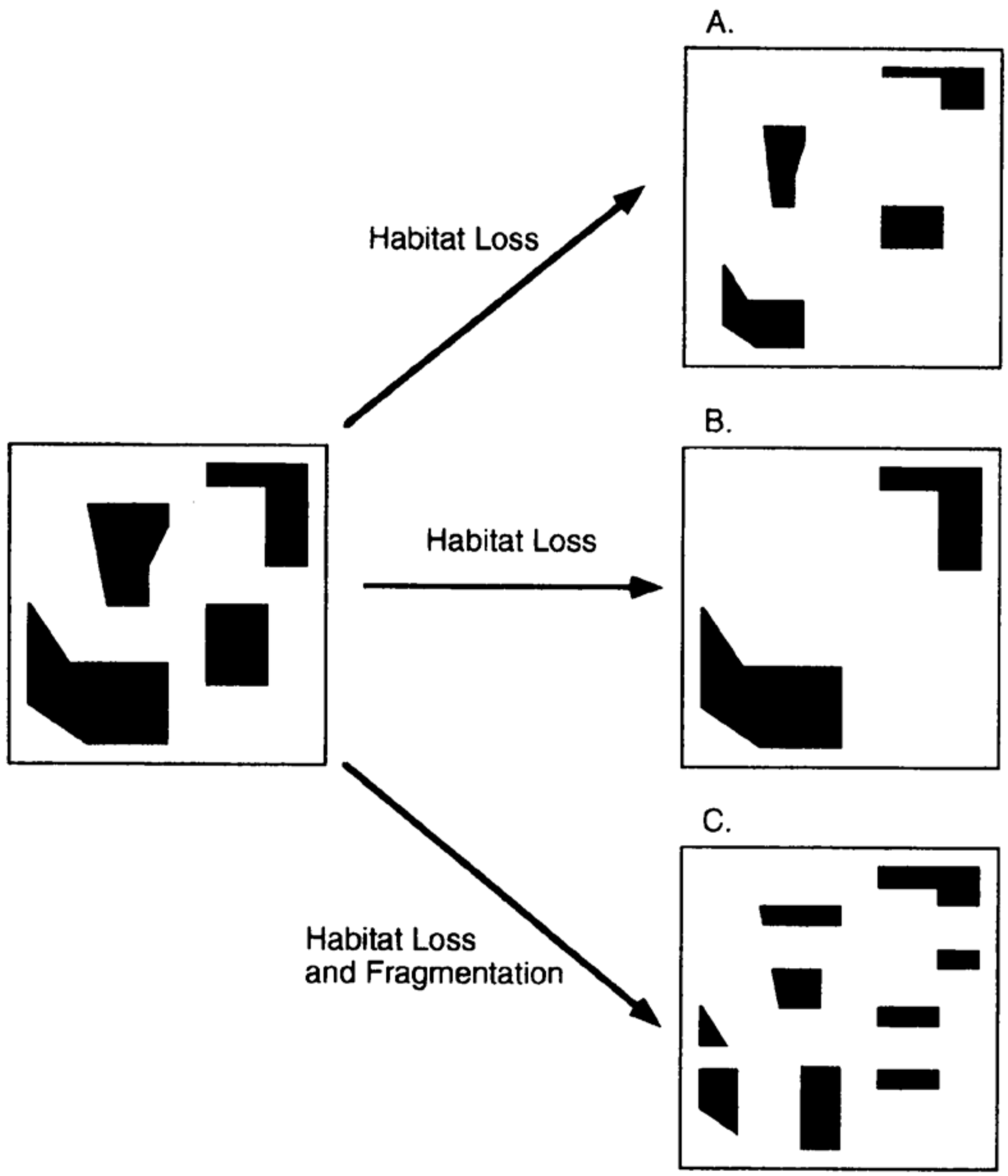

Figure 2.5: Effects of habitat loss and fragmentation on patch size and isolation (*Source*: Figure 1 in Fahrig 1997).

2.2.3 Effects of Habitat Fragmentation in General

Researchers are in agreement that under the broad definition of habitat fragmentation, explicitly including the breaking apart of and the loss of habitat, habitat fragmentation disrupts the connectivity (both structural and functional connectivity) of habitat patches, hindering the movement/dispersal of organisms; increases inter-patch distance (increases patch isolation), thereby increasing mortality during movement and reducing immigration or recolonization; reduces the size of remaining patches—populations in small patches are more likely to become extinct than those in large patches; and increases the amount of edge (Wilcox and Murphy 1985, Saunders et al. 1991, Ricklefs and Miller 2000, Cooper and Walters 2002, D'Eon 2002, Fahrig 2002, Noss and Daly 2006). Consequently, the effects of habitat fragmentation in general are reduced species abundance and richness (diversity) and decrease in population persistence (Noss et al. 2006). Responses to habitat fragmentation are variable across taxa (e.g., birds vs. amphibians) and also within a same taxon (e.g., different species of birds) (Robinson et al. 1993, Aizen and Feinsinger 1994, Margules et al. 1994, Villard and Taylor 1994).

Because habitat fragmentation per se and habitat loss usually occur together (Haila and Hanski 1984, Harrison and Fahrig 1995, Fahrig 1997, 1999, D'Eon 2002, Noss et al. 2006), some researchers (e.g., Simberloff 2000, Haila 2002) argue that most of these fragmentation effects can be explained by area effects: that a small area contains fewer number of species based on the theory of island biogeography (MacArthur and Wilson 1967). Therefore, Fahrig (1997, 2003) and others argue for the necessity of separating out the effect of habitat fragmentation per se. I agree with Fahrig and others that there is a need for measuring independent effect of habitat fragmentation per se to know how much fragmentation per se contributes to the consequent reduction in species abundance and richness (diversity) and in population persistence.

2.2.4 Independent Effects of Habitat Amount and Fragmentation Per Se

2.2.4.1 Empirical Studies

2.2.4.1.1 Habitat Amount More Important than Habitat Fragmentation

Westphal et al. (2003) found that while the total area of native vegetation around a site was the most important factor in determining the distribution of woodland bird species in the Mount Lofty Ranges, South Australia, landscape configuration was also important for many species. Most species responded positively to area-independent fragmentation, but the responses to mean patch isolation and mean patch shape were more variable (Westphal et al. 2003).

Radford et al. (2005) controlled habitat amount through the design of experiments. They compared landscapes with similar overall amounts of habitat but contrasting configuration (i.e., aggregated versus dispersed) for richness of woodland-dependent birds. They found habitat configuration exerted weaker influence than habitat cover (amount) but that it was not at all irrelevant in predicting species richness. The conclusion drawn from these two studies is that although habitat amount is more important for the distribution and richness of woodland birds than habitat fragmentation per se, fragmentation per se is still a relevant factor.

2.2.4.1.2 Landscape Composition More Important than Landscape Configuration

In general, the effects of landscape composition on ecological variables are large (Fahrig 2003). “Where landscape-studies have been conducted, large effects of landscape structure (especially landscape composition) have been found” (Fahrig 2005, p. 9).

Bennett et al. (2006) found that in studies that separate the independent effects of three categories of landscape mosaics (i.e., the extent of habitat, composition of the mosaic, and spatial configuration of elements) in agricultural landscapes, spatial configuration generally exerts less influence on biota than extent or composition.

Radford and Bennett (2007) studied the relative effects of habitat extent, habitat configuration, landscape composition, and geographical location on the occurrence of forest birds in agricultural landscapes. They found that although habitat configuration was important for fewer number of birds than habitat extent, for species with strong evidence of configuration effects, the effect of the independent measures of landscape configuration was very large.

2.2.4.1.3 Interaction Found

Betts et al. (2006b) examined the relative effects of habitat amount and habitat fragmentation per se, using two species of forest birds in forest-dominated landscapes. Landscape configuration (fragmentation per se) was shown to be important only for Ovenbird (*Seiurus aurocapillus*) and only when the amount of suitable habitat was low, claiming to be the first to report empirical evidence of the interaction between habitat amount and fragmentation. However, local habitat (e.g., hardwood basal area/ha) or landscape composition (e.g., amount of habitat within a 2000-m radius) variables, not landscape configuration variables, explained most variance in the occurrence of both species.

Cooper and Walters (2002) investigated the relative importance and the degree of the independent effects of woodland cover and fragmentation per se on Brown Treecreeper (*Climacteris picumnus*) distribution in a matrix of woodlands and pastures.

They found that woodland fragmentation per se was important at a broad scale (4.5-km radius) while both woodland cover and fragmentation per se were important at a finer scale. However, fragmentation per se was important only when < 20% of woodland cover remains at a given scale, suggesting a threshold effect and supporting the interaction effect. In sum, both studies found the interaction between habitat amount and fragmentation per se only when the amount of suitable habitat was low.

2.2.4.1.4 Interaction Not Found

Cushman and McGarigal (2003) used a combination of factorial analysis of variance and partial canonical variates analysis to quantify the relative importance of differences in mature forest area, fragmentation, and basin in influencing each response variable and community diversity overall. Unlike several other studies, they did not find that the relative strength of fragmentation increased as habitat area decreased. In other words, they did not find the negative interaction between habitat amount and fragmentation. They note that forest-dominated landscapes with a spatially complex seral mosaic may have influenced the results.

Parker and Mac Nally (2002) did not find an interaction between the effects of habitat loss and habitat fragmentation in the predicted fashion on the abundance and richness of grassland invertebrates (i.e., ants, beetles, dipterans, and hemipterans). They did not even find the general effects of habitat loss and fragmentation but a strong “edge-centre” difference and a temporal change in both richness and abundance.

Betts et al. (2007a) found little effect of patch size—used as the sole measure of fragmentation—on the occurrence of most species of birds they studied, regardless of the amount of habitat present at landscape extents. In other words, for most species, the interaction between patch size and habitat amount was not significant. They did find that that two out of 15 bird species were more likely to occur in large patches when the amount of habitat in a landscape was low.

In sum, these studies did not find the interaction between habitat amount and fragmentation per se. The effect of habitat fragmentation per se did not become stronger when the amount of suitable habitat was reduced below a certain percentage of a total landscape. The results seem to be applicable to forest birds in forest seral mosaics and grassland insects.

2.2.4.1.5 Habitat Fragmentation Equally Important but No Interaction

To assess the independent effect of forest configuration on the presence of forest bird species, Villard et al. (1999) regressed forest configuration metrics against forest cover and used the residuals in logistic regression models. They found that both forest cover and configuration were important predictors of species presence and that responses were species-specific. Also, they did not find any threshold amount of forest cover and configuration (fragmentation) on species presence, contradicting to other studies that did find the threshold. They maintain that although forest cover is an important predictor of these birds' presence, the effect of forest configuration is large enough to merit consideration in conservation strategies.

2.2.4.1.6 No Consistent Effect of Habitat Fragmentation from Marine Studies

Johnson and Heck (2006) claim to be the first marine study on the independent effect of fragmentation. While most terrestrial studies on the effect of fragmentation are conducted on forest birds, Johnson and Heck (2006) measured abundances of decapods and fishes and estimated secondary production of natural and artificial seagrass beds of varying sizes and spatial configuration. They found an inconsistent overall impact of patch size, patch shape, intra-patch location, and degree of isolation on macrofaunal community structure and secondary production estimates. Their data suggest that (1) the effects of habitat fragmentation are location-, time- and species-specific and that (2) fragmentation may have little impact on macrofaunal assemblages of seagrass meadows, whose patches ranging in size from 100 m^2.

Bell et al. (2001) did not find any consistent effect of habitat fragmentation on marine fauna from their Tampa Bay, Florida, experiments. Also, the infaunal polychaete, *Kinbergonuphis simony*, did not differ in their use of edge or core areas of seagrass patches. Bell et al. (2001) conclude that neither their review of the literature on fauna and seagrass patch size nor the data presented from their Tampa Bay studies suggest that habitat fragmentation has any consistent impact on fauna over the spatial scales that have been investigated.

2.2.4.1.7 Factors to Consider

Koper et al. (2007) caution that the relative strength of the independent effects of habitat fragmentation and habitat loss may depend on the method of analysis. They found that when the residuals of fragmentation were regressed on habitat amount, which is a standard method of obtaining the independent effect of fragmentation per se, the effect of fragmentation per se was found to be stronger than that of habitat amount. However, when they obtained the residuals of habitat amount regressed on fragmentation, they found that the effect of habitat amount was stronger than that of habitat fragmentation per se. The significance of Koper et al.'s (2007) study is that the relative strength of fragmentation may be influenced by the order in which the residuals are taken.

The landscape context may influence the relative effects of habitat amount and fragmentation (D'Eon and Glenn 2005). Cushman and McGarigal (2003) speculate that the landscape context of forest successional (i.e., seral) mosaics may be the possible reason for not finding the interaction between habitat amount and fragmentation per se.

2.2.4.1.8 Summary of Recent Empirical Studies

The most common finding of empirical studies of the independent effects of habitat amount (landscape composition) and fragmentation per se (landscape configuration) is that fragmentation per se is generally less important than habitat amount for the presence/absence and richness of species (Westphal et al. 2003, Radford et al. 2005, Bennett et al. 2006, Betts et al. 2006b, Radford and Bennett 2007). However, the effect of habitat fragmentation per se is strong on some species and/or only when the amount of habitat in a landscape is low (Cooper and Walters 2002, Betts et al. 2006b). Therefore, some studies conclude that the effect of fragmentation per se cannot be ignored (Villard et al. 1999, Radford et al. 2005, Radford and Bennett 2007).

The evidence for the existence of the interaction between habitat amount and fragmentation per se is inconclusive. Some studies found that the relative strength of fragmentation increased as habitat area decreased or that fragmentation was important only when habitat amount deceased beyond a certain threshold amount (Cooper and Walters 2002, Betts et al. 2006b). Other studies did not find the interaction between habitat amount and fragmentation (Villard et al. 1999, Parker and Mac Nally 2002, Cushman and McGarigal 2003).

Studies conducted in a marine environment find weak and variable effects of fragmentation per se on marine fauna (Bell et al. 2001, Johnson and Heck 2006). Also, the effect of fragmentation per se appears to be species-specific (Villard et al. 1999 for forest birds, Johnson and Heck 2006 for marine macrofauna). Characteristics of the marine environment, which in many ways different from the terrestrial environment, may influence the strength of the effect of habitat fragmentation per se on marine fauna.

Since it can be said that the concept of connectivity is the inverse of habitat fragmentation (i.e., the more fragmented habitat patches are, the less connected a landscape is for the particular organism of interest), with regards to my research questions of the first part, the literature review of the empirical studies of the independent effect of habitat fragmentation per se suggests that forest amount would likely be the most important predictor of the number of individuals of the selected forest bird species and that forest fragmentation per se would not be such an important factor in predicting the number of individuals. However, forest fragmentation per se may still be influential for some of the species and/or only when the amount of forest in a landscape is low. Actually, my study will serve as an additional observational study to the discussion of whether or not the interaction between habitat amount and fragmentation per se exists. Whether or not the effect of forest fragmentation per se increases as that of forest amount decreases remains to be seen. The effect of various land cover/use types on the number of individuals of the selected forest bird species will also be investigated; this question considers the effect of the surrounding landscape.

2.2.4.2 Simulation Studies

Fahrig (1997) tested the relative effects of habitat amount and spatial configuration (fragmentation) on population extinction (extinction probability and extinction time) using a spatially explicit simulation model. Results indicate that the effect of habitat loss is much larger than that of habitat arrangement. Therefore, Fahrig (1997) concludes that species conservation efforts should focus on preserving habitat first and habitat restoration second. Similarly, using simulation models, Fahrig (2002) found that in general, in more fragmented landscapes, more habitat was required for population persistence. Also, better performing models' prediction showed that habitat loss had a much larger effect than habitat fragmentation on population extinction.

Using simulation models, Fahrig (2001) investigated the relative effects of the four factors (i.e., reproductive rate of the organism, rate of emigration of the organism from habitat, habitat pattern in the landscape, and matrix quality [survival rate of the organism in non-habitat areas]) thought to influence the relationship between habitat loss and the probability of population extinction. Among these four factors, reproductive rate had the largest potential effect on a threshold amount of habitat loss at which the probability of population extinction drastically changes by a small additional loss of habitat; habitat pattern had a very small predicted effect.

Flather and Beavers (2002) studied the relative effects of habitat amount and habitat arrangement on the population size of a hypothetical species using a discrete reaction-diffusion model. Overall, the effect of habitat amount was much larger than that of habitat arrangement. They did find that the effect of fragmentation increased when the percentage of habitat amount decreased to 30-50% of the landscape (the threshold habitat amount).

The results from these simulation models suggest that the effect of habitat loss is much larger than that of habitat arrangement on population size and extinction. The evidence for the effect of spatial arrangement of habitat on the threshold amount of habitat below which the probability of population extinction drastically increases is equivocal. Flather and Beavers' (2002) study found the interaction between habitat amount and fragmentation; Fahrig's (2001) study found little interaction. Reproductive rate of the simulated organism was much more important for the threshold amount of habitat than habitat pattern (Fahrig 2001).

The much larger importance of habitat loss than habitat arrangement supports the kind of conservation planning policy that would focus on preserving existing habitat first and habitat restoration next. The results of the simulation studies imply the need to preserve existing suitable habitat before making connections among them or thinking about their spatial configuration. The result that species-specific traits such as the reproductive rate of an organism was more important (than the spatial configuration of habitat) for the threshold amount of habitat (Fahrig 2001) cautions the one-plan-fits-all (the species) type of approach to conservation planning. Certainly, species respond to fragmentation variably—among species of different taxa and even among the species of a same taxon.

2.2.4.3 General Conclusions

Both empirical and simulation studies overwhelmingly find that the effect of habitat amount (loss) is much larger than that of habitat fragmentation per se on the abundance and distribution of species (Fahrig 2002, Flather and Beavers 2002, Westphal et al. 2003, Radford et al. 2005, Bennett et al. 2006, Betts et al. 2006b, Radford and Bennett 2007). The caveat here is that most of the empirical studies are based on forest birds. However, few studies (e.g., Mac Nally and Brown 2001) using other taxonomic groups have found the same result, indicating that this general conclusion may be more generally applicable. Also, whether or not habitat patches have high contrast to the surrounding matrix seems to influence the results (Cushman and McGarigal 2003). In some studies (e.g., Villard et al. 1999, Radford et al. 2005, Radford and Bennett 2007), however, the effect of habitat fragmentation per se is strong enough to merit consideration in conservation planning.

The results are inconclusive for the presence of the interaction between habitat amount and fragmentation. Some studies (e.g., Cooper and Walters 2002, Betts et al. 2006b) find a stronger effect of fragmentation per se when the amount of suitable habitat is low; others (e.g., Villard et al. 1999, Parker and Mac Nally 2002, Cushman and McGarigal 2003) do not. Another conclusion is that the response to fragmentation per se and the strength of its effect are variable among species and even among species of the same taxon such as birds (Villard et al. 1999, Betts et al. 2006b, Johnson and Heck 2006).

2.2.4.4 Ways to Achieve Independence

Since habitat fragmentation per se often accompanies habitat loss, to distinguish the independent effect of fragmentation per se, we need methods to separate its effect from the confounded effects of habitat loss and fragmentation per se. We can statistically isolate the effect of fragmentation per se. McGarigal and McComb (1995) and Trzcinski et al. (1999) used a multivariate method—for example, by generating a fragmentation index (independent of forest cover), using principal component analysis (PCA) from the measures of mean forest patch size, number of forest patches, and total forest edge. Another statistical approach is using covariates to remove correlation. For example, Flather and Bevers (2002) removed the covariation with habitat amount. Cushman and McGarigal (2003) used partial canonical variates analysis. Similarly, Villard et al. (1999) regressed habitat configuration metrics against habitat amount and used the residuals in the following statistical models. Trzcinski et al. (1999) actually combined the both method: they first generated a single measure of fragmentation per se by PCA of various landscape configuration metrics (a multivariate method); then, regressed the index (the principal component) against forest cover, and used the residuals as a measure of forest fragmentation to completely remove the remaining correlation (found to be non-significant).

Another way to isolate the effect of habitat fragmentation per se is by appropriate experiment design. For example, Betts et al. (2006b) used a stratified sampling design that reduces the confounding of habitat amount and fragmentation variables. Similarly, Radford et al. (2005) compared landscapes with similar overall amounts of habitat but contrasting configuration (i.e., aggregated versus dispersed). Mac Nally and Brown (2001) examined the effect of fragmentation on biodiversity of terrestrial reptiles in south-eastern Australia. They used two sets of four size classes of patches: one set is in a "fragmented" landscape and the other is embedded in large contiguous forests to isolate fragmentation effects from area effects. Fragmentation did not have significant effect on total numbers and richness. However, species response to fragmentation in terms of occurrence and abundance was variable.

2.2.4.5 Discussion

Realistically, it is difficult to distinguish habitat fragmentation per se and the loss of habitat in landscape transformations, particularly at a patch scale, because these two processes often occur together (Haila and Hanski 1984, Harrison and Fahrig 1995, Fahrig 1997, 1999, D'Eon 2002, Noss et al. 2006). Therefore, many previous studies (e.g., Wilcox and Murphy 1985, Noss 1991, Robinson et al. 1995, Schumaker 1996, van den Berg et al. 2001, Hovel 2003) that reported the effects of habitat fragmentation (or, so they claimed) actually reported the confounded effects of both fragmentation per se and habitat loss on species abundance, richness, population persistence, or dispersal success. Few studies (e.g., McGarigal and McComb 1995, Fahrig 1997, Trzcinski et al. 1999, Villard et al. 1999, Flather and Bevers 2002) have reported the independent effect of fragmentation per se (i.e., the breaking apart of habitat after controlling for habitat amount). These studies predominantly found much smaller effect of fragmentation per se, as compared to that of habitat loss, on species presence/absence (Trzcinski et al. 1999), abundance (McGarigal and McComb 1995, Flather and Bevers 2002), or population survival (Fahrig 1997). (Villard et al. [1999] found equally strong effects of fragmentation per se.) Also, the independent effect of habitat fragmentation per se was both positive and negative (McGarigal and McComb 1995, Trzcinski et al. 1999, Villard et al. 1999). Fahrig (1997) and Trzcinski et al. (1999) in particular found that fragmentation per se did not have a predicted strong effect on population persistence and species presence/absence even when its effect was expected to be strong (i.e., when the percentage of habitat on a landscape was low), as suggested by McLellen et al. (1986), Andrén (1994), Fahrig (1998), and Flather and Bevers (2002). The existence of the threshold amount of habitat exists below which the effect of fragmentation per se increases and the relative strength of the effect of fragmentation per se are contested (D'Eon 2002). The caveat is that all the studies except for Fahrig (1997) and Flather and Bevers (2002), which are simulation studies, dealt with forest birds. It would be interesting to know whether or not the general conclusion that the effect of habitat loss is much larger than that of fragmentation per se holds true for other taxa. There are few studies of the independent effects of habitat amount and fragmentation per se on the richness and abundance of terrestrial reptiles (Mac Nally and Brown 2001), grassland invertebrates (Parker and Mac Nally 2002), and marine macrofauna (Johnson and Heck 2006). It is to be seen that the results will be more generally applicable to other taxonomic groups.

Because of the difficulty in isolating the effect of habitat fragmentation per se from that of habitat loss, which often co-occurs, there have been few studies (especially, empirical studies) that clearly showed the independent effect of habitat fragmentation per se on the abundance and distribution of species. Although the evidence is mounting for a much stronger effect of habitat amount than habitat fragmentation per se, the jury is still out on the strength of the effect of fragmentation per se when the amount of suitable habitat in a landscape is low. More studies are needed to examine the relative importance of habitat amount and fragmentation per se.

As for my research questions, based on the literature review, I would expect the number of individuals of the selected forest bird species, at least for a couple of species, to show a threshold response to forest cover (amount). I would also expect to find overall much larger effect of forest cover than forest fragmentation per se on forest bird abundance. Whether or not the effect of forest fragmentation per se becomes stronger as the amount of forest decreases or when forest cover is reduced below the threshold remains to be seen. Species response to fragmentation per se is likely variable.

2.2.4.6 Conservation Planning Implications

If fragmentation per se can have a large effect on the abundance and distribution of species, alteration of habitat spatial configuration (independent of habitat amount) will be an effective tool for species conservation. On the other hand, if the effects of fragmentation per se are small, this is a limited option (Fahrig 2002). Similarly, effective conservation and management strategies should be different for the species that are sensitive to the overall amount of habitat as to the species that are sensitive to habitat fragmentation per se (Collinge 2009).

Based on the empirical and simulation studies that investigated the independent effects of habitat amount and fragmentation per se, Haila (1986), Harrison and Fahrig (1995), Fahrig (1997, 1998, 1999, 2001, 2002), Trzcinski et al. (1999), McGarigal and Cushman (2002), and Schmiegelow and Mönkkönen (2002) conclude that habitat amount is much more important than its spatial configuration for species presence/absence, richness/abundance or population survival, and recommend that species conservation efforts should focus on habitat preservation and restoration, securing a sufficient amount of habitat first before considering their spatial configuration and/or connecting them with corridors.

However, rarely do we find empirical studies that can measure the effect of fragmentation per se independently of that of habitat amount (percentage) on many species at a landscape scale. Moreover, we may also need information on dispersal and survival in various types of habitat for each species to truly show the independent effect of habitat fragmentation per se. There is deficiency of this kind of data.

With regard to my research questions, along with examining the existence of the interaction between forest amount and fragmentation per se, it would be interesting to see whether or not there is a threshold amount of forest cover for the number of individuals of the selected forest bird species. If there is a threshold of forest amount below which the number of individuals suddenly declines, the threshold may aid in taking a proactive conservation action before the amount of forest in a region is reduced below that level or serve as a useful target of restoring the amount of forest to that level. This would translate to conservation planning actions such as prioritizing land management or acquisition options, and targeting areas for restoration.

2.2.4.7 Conclusions

I echo Fahrig's (1999) and Trzcinski et al's (1999) concern that the danger in confounding the effect of habitat fragmentation per se with that of habitat loss, or worse yet confusing the effect of habitat loss with that of fragmentation per se is that it could lead to an erroneous conclusion that it is acceptable to lose habitat as far as the remaining habitat patches can be spatially arranged in a way which would compensate the effects of the lost habitat area. There are those (e.g., Kareiva and Wennergren 1995, With and King 2001) who argue that landscape configuration can mitigate the effects of habitat loss and enhance population persistence in fragmented landscapes. The studies (McGarigal and McComb 1995, Fahrig 1997, 2002, Trzcinski et al. 1999, Flather and Bevers 2002, Westphal et al. 2003, Radford et al. 2005, Betts et al. 2006b) which separated the effect of fragmentation per se from that of habitat loss indicate otherwise: that habitat loss exerts much larger influence on species presence/absence, abundance, richness, or population survival than fragmentation per se, and therefore, the areas lost cannot be easily compensated by the spatial arrangement of the remaining habitat patches.

If the effect of forest fragmentation per se becomes significant/stronger when the threshold amount of forest is passed, this underscores the importance of the spatial configuration of forest patches in conservation planning, lending support to policies that would increase connectivity between them or trying to locate them in close proximity. On the other hand, if there is weak or no interaction between forest amount and fragmentation per se, this would support a policy—if the goal of the landscape plan is to provide enough habitat to be able to sustain the populations of these species of forest birds—which recommends the preservation of existing forests first, and second, restoring them. Protecting and restoring forest habitats should be the priority, not developing corridors or mulling over the spatial configuration of these forest habitats.

I agree with Fahrig and others' argument that if the effect of habitat fragmentation per se is not clearly distinguished from that of habitat loss, erroneous conclusions can be reached and conservation planning recommendations based on these misleading conclusions could have devastating effects on the species of interest. Fortunately, there are methodological ways to independently measure the effect of habitat fragmentation. The merit in taking this extra care to insure that we are measuring habitat fragmentation effects separately from habitat loss effects is large and the effort can result in conservation and management strategies that are more appropriate for protecting the target species.

2.2.5 Effects of Habitat Fragmentation Per Se Must Be Studied at a Landscape Scale

2.2.5.1 What is a Landscape-scale Study?

According to Fahrig (2005), a landscape-scale study requires researchers to compare multiple landscapes with different structures; whereas, a patch-scale study uses the information from only one landscape. To be able to answer how landscape structure affects (the processes that determine) the abundance and/or distribution of organisms, the response variable (abundance/distribution/process) must be compared across different landscapes with various structures (Brennan et al. 2002, Fahrig 2005). As seen in Figure 2.6, in a patch-scale study, each data point represents the information from a single patch, and only one landscape is studied. On the other hand, in a landscape-scale study, each data point represents the information from an individual landscape, and multiple landscapes with different structures are studied. The appropriate size of a landscape for a landscape-scale study depends on the scale at which the response variable operates—for example, a daily movement range of an organism or the maximum between-population dispersal distance of the amphibian species of interest (Fahrig 2005). A landscape-scale study can be conducted at any level of biological organization (i.e., individual, population, community, or ecosystem level) (Fahrig 2005).

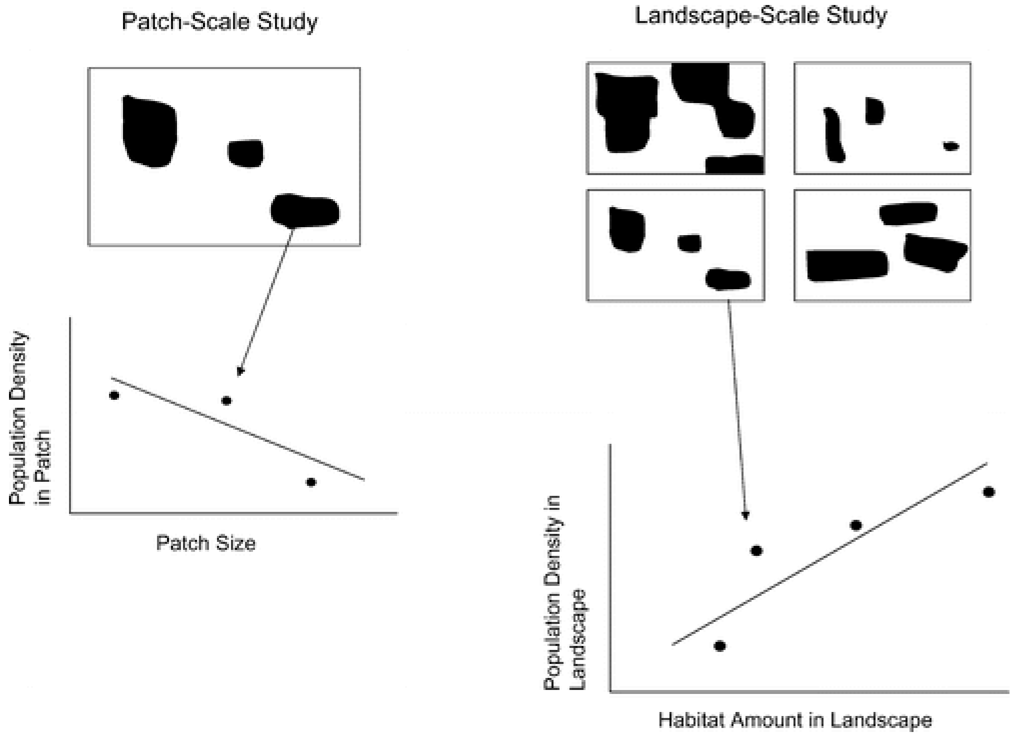

Figure 2.6: Comparison between a patch- and landscape-scale study. (A) Patch-scale study: each observation represents the information from a single patch (black areas). Only one landscape is studied, so sample size for landscape-scale inferences is one. (B) Landscape-scale study: each observation represents the information from a single landscape. Multiple landscapes, with different structures, are studied. Here, sample size for landscape-scale inferences is four (*Source*: Figure 4 in Fahrig 2003).

2.2.5.2 Why do Landscape-scale Studies Necessary? What are Their Advantages over Patch-scale Studies?

As noted in Figure 2.6, the problem with a patch-scale study is that the sample size for landscape-scale references is one. Therefore, the results of the study cannot be generalized to other landscapes. On the other hand, a landscape-scale study allows landscape-scale inferences. Bennett et al. (2006) argue that to be able to infer landscape-scale influence, sampling must encompass multiple land uses and elements within a landscape to represent the "whole" landscape mosaic, and be replicated across multiple landscapes. This means that a landscape needs to be the unit of replication to examine the effects of land mosaic characteristics (e.g., the extent of habitat, composition of the mosaic, and spatial configuration of elements) on the abundance and distribution of organisms. Radford and Bennett (2007) also argue for the need to conduct landscape-scale studies (considering whole mosaics) not patch-scale studies (individual patches) to be able to understand the effects of the composition and heterogeneity of land mosaics.

2.2.5.3 Confusion between a Landscape "Scale" and a Landscape "Level"

The terminologies used in FRAGSTATS (McGarigal et al. 2002)—as in Patch-Class-Landscape "levels" when applying landscape metrics—add confusion to this issue. Patch-level metrics are defined for individual patches, and characterize the spatial character (e.g., size, perimeter, and shape) and the context of patches. Aggregate properties of patches are measured at two levels: class and landscape. Class-level metrics are integrated over all the patches of a given type (class). They are used to study the amount and distribution of a particular patch type, and thus useful in studying habitat fragmentation (McGarigal et al. 2002). Landscape-level metrics are integrated over all patch types or classes over the full extent of the data (i.e., the entire landscape). Their primary application is the study of the relationship between the structure (i.e., composition and configuration) of the entire landscape mosaic and ecological processes (McGarigal et al. 2002). The patch-class-landscape level metrics are FRAGSTATS-specific terminologies and they serve as a useful, organizational framework of quantifying landscape structure. Therefore, I will accept and use them in my methods and writing. However, the usages such as patch-level and landscape-level metrics should not be confused with the general usage of landscape-scale (level) studies.

Another use of "landscape" and "landscape-scale" comes from the studies that conduct a multi-scale analysis. They equate the term "landscape" simply to a large spatial extent (around a sample point) as compared to the patch where the sampling point is located—and refer this as a “patch” or “local” scale (e.g., Fletcher and Koford 2002, Crozier and Niemi 2003, Melles et al. 2003, Betts et al. 2007a). The problem is that the decision to call a certain size of an area a landscape seems arbitrary and also is relative to the sizes of all the measurements in the same study. For example, Betts et al. (2007a) use >= 500-m radius (of a circle) as a landscape; whereas, Crozier and Niemi (2003) define 1-km^2 area around each sample point as landscape. Betts et al. (2007a) used four spatial extents (150-m, 500-m, 1000-m, and 2000-m radius) in their multi-scale analysis and called areas with >= 500-m radius as landscapes and referred to the smallest circle with 150-m radius as a local scale. Similarly, Melles et al. (2003) differentiated local- and landscape-scale by the extent not the number of replication at that scale. They called characteristics within 1000 m as landscape-level features and those within 50 m as local-scale habitat measures. Ecologically meaningful landscape size should correspond to the scale at which a response variable responds to landscape structure (Wiens 1999, Fahrig 2005). For example, Betts et al. (2007a) state that their choice of the study area (extent) is based on the size of territories, dispersal distance, and extraterritorial movements of the focal bird species.

Furthermore, researchers (e.g., Graham and Blake 2001, Cushman and McGarigal 2003, Radford et al. 2005, Bennett et al. 2006, Radford and Bennett 2007) use the term “level” interchangeably with “scale”—both in the context of the spatial extent and a landscape-scale study. The FRAGSTATS’ term “landscape-level” and the use of “landscape” to suggest a large spatial extent are all related but should not be confused with the use in the definition of landscape-scale studies. These ambiguities can be resolved by accurate and explicit definitions of each term.

2.2.5.4 Hindrance to Landscape-scale Studies

There is a lack of landscape-scale studies in the literature (McGarigal and Cushman 2002, Fahrig 2005). This is because conducting manipulative experimentation (just as one would do in a controlled laboratory) at a landscape scale is almost impossible due to the number of replication of treatments required to achieve statistical significance (Fahrig 2005). The funding to conduct such true landscape-scale studies is very difficult to obtain with today's limited resources and budget. Researchers can remedy this problem by engaging in what is called a "mensurative experiment" (Hurlbert 1984, Hargrove and Pickering 1992, McGarigal and Cushman 2002) by carefully choosing landscapes that cover a range of structures (e.g., Trzcinski et al. 1999, Cushman and McGarigal 2003, Betts et al. 2006b). In managed forest, collaboration between researchers and natural resource managers is the key to conducting landscape-scale studies (D'Eon 2002).

2.2.5.5 Habitat Fragmentation Per Se Must Be Studied at a Landscape Scale

Fragmentation per se is best conceived as a landscape-scale process not a patch-scale process (Fahrig 1999). As seen in Figure 2.6, a landscape-scale study involves the comparison of many landscapes; a patch-scale study involves studying each patch in only one landscape. Too often fragmentation effects on population density or distribution are concluded by studies conducted at a patch scale; these fragmentation effects are actually due to patch size and isolation (distance among patches) (McGarigal and McComb 1995, Fahrig 1999, 2003). As seen in Figure 2.5, habitat loss alone at a “landscape” scale can account for decrease in “patch” size (A) and increase in “patch” isolation (A and B). When the number of patches increases by the breaking apart of habitat patches, both habitat loss and fragmentation per se are involved in the decreasing size and increasing isolation of habitat patches (C). Case C is a typical example of fragmentation including habitat loss. When the number of habitat patches remains constant, habitat loss alone can increase patch isolation by the reduction in area of each of the habitat patches (A). When some of the entire patches are removed (Forman [1995] calls this attrition), isolation increases but fragmentation per se (the breaking apart of habitat, independent of habitat loss) actually decreases because there are fewer patches (B). In this case, isolation effects are caused by habitat loss alone.

The cases A and B may be confused with habitat fragmentation if studies are conducted at a patch scale. What seems as if the effect of fragmentation is actually the effects of reduced patch size and/or increased patch isolation. This confusion and misunderstanding are caused by the focus on each patch. Patch size effects or patch isolation effects should be studied at a landscape scale not at a patch scale. Documentation of the effect of habitat fragmentation per se therefore requires the true landscape-scale study.

2.3 Connectivity from a Landscape Ecological Perspective

2.3.1 Introduction

Understanding the relationship between landscape pattern and process is the central research theme in landscape ecology (Turner 1989, 2005, Turner et al. 2001, Fortin and Agrawal 2005). The principles and theories of landscape ecology can be used to balance human and non-human needs in conservation and regional development plans (Flores et al. 1998, Ahern 1999, Zipperer et al. 2000, Li et al. 2005a, Yli-Pelkonen and Niemela 2005). Among the principles and theories of landscape ecology, many researchers (e.g., Zipperer et al. 2000, Moilanen and Hanski 2001, Tischendorf and Fahrig 2001, Wiens 2005) have identified connectivity as an important research topic in landscape ecology. For example, connectivity of habitat is critical for organisms to move, migrate, and disperse between habitat patches, which facilitates a gene flow and helps maintain physically separated populations (Bennett 1998, Soulé et al. 2004). Because the issue of connectivity resides in the interface between landscape pattern and process, it is central to the landscape ecology research. Since connectivity—especially, functional connectivity—is influenced by the interaction of the target ecological process with the landscape, connectivity is said to have an emergent characteristic (Green 1993). Considering the concept of connectivity for target ecological processes helps landscape planners decide, for example, how to best place green spaces in urban environments (Flores et al. 1998, Zipperer et al. 2000). Different levels of connectivity provide a useful conceptual framework to organize green spaces across scales and challenge landscape planners to achieve connectivity in this manner.

2.3.2 Definition of Connectivity

Connectivity is usually conceived in terms of the movement of matter and energy across landscapes (Forman and Godron 1986, Forman 1995). Forman (1995) defines connectivity as spatial continuity of a habitat or cover type across a landscape (i.e., no breaks). This definition deals only with physical connectedness and lacks a particular species or ecological process perspective (i.e., organism-centered perspective) as discussed later. Tischendorf and Fahrig (2000b) argue for landscape-wise connectivity defined as the degree to which a landscape facilitates or impedes movement of organisms among resource patches. However, the rate of movement is difficult to measure. Turner et al. (2001) take a broader view to include other ecological processes than the movement/dispersal of organisms in the definition of connectivity. They state that connectivity is defined by the relationships between landscape structure and the ecological process of interest "that connects adjacent sites" (p. 154).

Broadly speaking, connectivity can be classified into two types: structural (physical) and functional connectivity. While structural connectivity concerns physical connectedness of habitat patches (With and King 1997, Tischendorf and Fahrig 2000a), functional connectivity is connectivity from the point of view of organisms interacting with the landscape—how they perceive and respond to landscape structure. Functional connectivity can also be defined as connectivity that can support a particular function such as being able to walk to go grocery shopping in a New Urbanist type of development. The difference between structural and functional connectivity can be illustrated by the following example. Structural connectivity of forest patches may be same for forest birds and small mammals but functional connectivity appears different for these two species. The same spatial configuration of habitat patches may appear to be more connected for forest birds that are more mobile than small mammals. Therefore, supposing that these two species can utilize the same habitat, the same landscape appears more functionally connected for forest birds than small mammals.

2.3.3 Measures of Connectivity

According to the definitions, connectivity can be measured structurally and functionally. Structural connectivity is a measure of physical connectedness of patches (With and King 1997, Tischendorf and Fahrig 2000a). Various landscape pattern indices have been proposed to measure structural connectivity (Goodwin 2003, Calabrese and Fagan 2004). For example, structural connectivity metrics in FRAGSTATS, a computer software program to compute various landscape metrics for categorical map patterns (McGarigal et al. 2002), include correlation length, the patch cohesion index, the connectance index, and so on. They represent structural connectivity in a slightly different way but essentially measure the contiguity of patches or like-cells (McGarigal et al. 2002).

As discussed above, functional connectivity varies from one ecological process to another. Therefore, it is said to have an emergent characteristic (Green 1993). Functional connectivity is decided by the way ecological processes interact with the landscape. For example, the way pest outbreaks propagate through a landscape may depend on the distribution of host species on the landscape and the moving capability of the pest species. FRAGSTATS offers a couple of metrics of functional connectivity such as the connectance index and the traversability index (not yet implemented). They require species-specific parameters such as dispersal distance and movement resistance by the landscape matrix. More empirical data for each concerned species (e.g., endangered species) are needed to determine these parameters. Expert opinions are also used to input the parameters.

Tischendorf and Fahrig (2000b) argue that connectivity should be measured according to its original definition: the degree to which a landscape facilitates or impedes movement of organisms among resource patches (Taylor et al. 1993, With et al. 1997). According to the authors, landscape connectivity is quantified by measuring species' movement among habitat patches or the rate of immigration into the focal patch(es). Only four out of 33 articles they reviewed accurately measured connectivity by dispersal success or search time (later criticized as not accounting for matrix mortality); the rest used surrogate measures (e.g., species' presence/absence data) that are, in their opinion, not accurate representation of connectivity (Tischendorf and Fahrig 2000b). In the review, Tischendorf and Fahrig (2000b) also found studies that associated the existence of corridors with increase in connectivity. They argue that corridors can facilitate species' movement among habitat patches but do not determine landscape connectivity. In other words, the connecting function of corridors does not always lead to increased, successful movement of organisms among resource patches (Tischendorf and Fahrig 2000b).

2.3.4 Inadequacy of Existing Landscape Metrics to Infer Ecological Functions

Numerous landscape indices have been proposed to represent and analyze landscape pattern (Plotnick et al. 1993, McGarigal et al. 2002). The landscape pattern indices derived from raster maps in particular (as compared to vector maps) are called landscape metrics (McGarigal et al. 2002). Landscape metrics provide quantitative information depicting the characteristics of various landscape patterns. Landscape metrics are expected to work as surrogates for landscape functional variables that are complex and often difficult to measure (Leitão et al. 2006). What is often missing, however, is the evidence for a clear relationship between landscape metrics and ecological processes

(With and King 1997, Corry 2005, Corry and Nassauer 2005, Li et al. 2005b). Granted that landscape pattern can be fairly accurately quantified by landscape metrics, how useful are these landscape metrics for inferring ecological processes?

Existing landscape metrics are not adequate for inferring ecological functions (Wu and Hobbs 2002, Corry 2005, Corry and Nassauer 2005, Li et al. 2005b). In other words, it is difficult to translate landscape metrics values to ecological meanings. For example, what landscape metrics quantify may be different from what is relevant to target ecological functions (Corry and Nassauer 2005). A set of (ideally) orthogonal metrics should be used to understand the big picture and metrics values should be taken relative to each other, relating to each landscape/plan, not as absolute (Leitão et al. 2006). More studies are needed to explicitly link landscape metrics to ecological processes (Corry 2005, Corry and Nassauer 2005, Li et al. 2005b, Leitão et al. 2006).

Moreover, landscape metrics have not traditionally been applied to human-dominated landscapes such as agricultural and urban landscapes, where remaining wildlife habitats are highly fragmented (Corry 2005, Corry and Nassauer 2005). When applied, at a very high resolution (3 m), to agricultural landscapes where good habitats are small (0.16-10.64 ha) and occur as linear features (e.g., roadside or fencerow habitats), configuration indices such as the contagion index, the aggregation index, and mean nearest neighbor distance did not provide valid information (Corry 2005, Corry and Nassauer 2005). In this situation, Corry and Nassauer (2005) recommend the use of indices that measure landscape composition to quantify a pattern difference (for example, among different landscape scenarios) but not ecological function. Also, at a very fine scale (~ 1 m), little is known about the behavior of landscape metrics although this fine scale is relevant for many designed landscapes in urban areas and for cultural landscapes such as settlement patterns and small remnant habitat patches in cities (Corry and Nassauer 2005).

Furthermore, we lack the knowledge of the ecological functions of small patches in human-dominated landscapes (Corry and Nassauer 2002). Corry and Nassauer (2002) argue that the values of small patches in agricultural landscapes are underestimated and not well studied. Much can be learned by investigating where to locate small habitat patches for increasing ecological functions (Corry and Nassauer 2002).

In landscape planning, landscape metrics are a useful tool for evaluating alternative plans and design and making informed decisions among them. Landscape metrics are used to quantify landscape structure, landscape change over time, landscape change before and after a landscape plan implementation, and to evaluate alternative plans and designs (Gustafson 1998, Leitão and Ahern 2002, Leitão et al. 2006). The question is: Can landscape metrics effectively predict ecological functions? The usefulness of landscape metrics for inferring ecological processes is still limited (With and King 1997, Turner et al. 2001, Wu and Hobbs 2002, Li et al. 2005b) and evidence is lacking to effectively link landscape metrics to ecological processes particularly in human-dominated landscapes at a fine scale (Corry 2005, Corry and Nassauer 2002, 2005). We need more studies that clearly link landscape metrics with specific ecological processes.

2.3.5 Threshold of Habitat Connectivity

Habitat connectivity has a threshold phenomenon and depends on the abundance and spatial arrangement of the habitat, as well as the movement or dispersal characteristics of the organism (Wiens et al. 1997, With and King 1997, McIntyre and Wiens 1999, Turner et al. 2001). Turner et al. (2001) define a habitat connectivity threshold as “a point at which the habitat suddenly becomes either connected or disconnected” (p. 234). This sudden change of the status from being connected to not connected is the key feature of the threshold of habitat connectivity.

Using simple random maps and the nearest-neighbor rule to define a patch, Gardner and O’Neill (1991) found that when the habitat (as to non-habitat) consists of about 60% of a landscape, a single contiguous cluster (i.e., a percolating cluster) forms across the map. This percentage of habitat, above which a percolating cluster forms, is called the critical threshold of connectivity, or the percolation threshold. Thus, the threshold of habitat connectivity for simple random landscapes with the four-neighbor rule is about 60%.

Although the percolation theory based on neutral landscape models (NLMs) (With 1997), using the four-neighbor rule, suggests 60% of the initial habitat (as opposed to non-habitat) as the threshold amount (percentage) of habitat before the percolating cluster breaks down, empirical studies (e.g., Lande 1987, 1988, Haila 1990, Haila et al. 1993, Andrén 1994) have found that 10-30% of the remaining habitat to be the threshold of habitat connectivity for small mammals and birds. Drastic changes in species’ response (or phenomena/other ecological processes such as the way disturbance and pathogens spread across a landscape) can occur when the available habitat is reduced below the threshold of the original habitat (Turner 1989, Turner et al. 1989, Gardner and O’Neill 1991, Gustafson and Parker 1992).

The threshold of habitat connectivity is affected by the amount of habitat (Gardner and O'Neill 1991, Andrén 1994, Hanski et al. 1996, Fahrig 1997, 1998, 2001, 2002, Jansson and Angelstam 1999, With and King 1999, Radford and Bennett 2004), the spatial configuration of the habitat (Andrén 1994, Hill and Caswell 1999, With and King 1999), the movement or dispersal characteristics of the organism/particular ecological process (Plotnick and Gardner 1993, Pearson et al. 1996, Turner et al. 2001, Radford and Bennett 2004), and the surrounding landscape matrix (Andrén 1994, Pearson et al. 1996, Mönkkönen and Reunanen 1999, With and King 1999, Fahrig 2001, Nikolakaki 2004, Radford and Bennett 2004). In sum, exactly where the threshold is depends on the organism, the amount of habitat, the spatial clustering of the habitat, and the nature of the matrix; different species may perceive different thresholds in the same landscape (Pearson et al. 1996, With 1997, With and King 1999).

2.3.6 Implications of the Application of the Critical Threshold of Habitat Connectivity to Conservation Planning

The caveat in applying a generalized critical threshold (one threshold value) to all the species is that each species has different habitat requirements and life history traits (Mönkkönen and Reunanen 1999). For example, species differ in their dispersal abilities, specific habitat requirements, and operational scales (e.g., large difference in home range sizes). Also, "[t]he applicable critical threshold depends also on the goal of the management, whether it is to preserve all species or just some proportion of them" (Mönkkönen and Reunanen 1999). However, identifying critical thresholds in habitat connectivity helps natural resource managers and landscape planners to design nature reserves and manage landscapes for biodiversity (Mönkkönen and Reunanen 1999).

The danger in applying a generalized critical threshold level for management of species and population survival is that (1) it is based on the requirement of more common and widespread species and therefore, (2) more sensitive species will likely go extinct before the generalized threshold is reached (Andrén 1999, Mönkkönen and Reunanen 1999). Thus, the requirements of more sensitive species should also be included in landscape planning (Mönkkönen and Reunanen 1999). For example, because forest-interior breeding bird species are area-sensitive and/or fragmentation-sensitive, they are likely to be affected first by the decline of the amount of forest cover and/or the increase in distance between forest patches (Whitcomb et al. 1981, Robbins et al. 1989, Fahrig 1999, Lee et al. 2002). Therefore, if forest-interior bird species are included in the indicators to measure the changes in forest amount and spatial configuration, the findings of the study would be helpful in identifying the threshold amount of forest, if any, to protect these species and other forest-associated species.

The significance of the threshold concept to species conservation is that if the extinction threshold exists, then a small additional loss of habitat near the threshold will have a large effect on population survival probability (Fahrig 2001). This is why it is critical to predict the threshold amount of habitat before it is reached so that resource managers and planners can take measures to plan for, mitigate, or compensate habitat loss and prevent extinction (Fahrig 2001). The review of the literature on the effects of impervious surfaces on stream quality shows that a similar threshold appears to exist for the percentage of impervious surfaces in a watershed before water/stream quality starts to degrade (Schueler 1994, Arnold and Gibbons 1996, Booth and Jackson 1997, May et al. 1997, Wang et al. 2001, Miltner et al. 2004). Therefore, the study of the relationship between forest bird and habitat (chapter 3) could be adapted to relate other ecological processes (e.g., hydrology) to spatial patterns of land use and thresholds.

Also, due to the slow response time of some species to the changes in habitat amount, we may not detect a problem with a population until the habitat is reduced well passed the threshold (Turner et al. 1994, Brooks et al. 1999, Balmford et al. 2003). Therefore, “we need to predict extinction thresholds for species before declines are observed, in order to avoid population decline and extinction due to habitat loss” (Fahrig 2001). This provides another supporting argument for a proactive planning to protect/restore habitat before the threshold is crossed.

2.3.7 Spatial Configuration of Habitat and Metapopulations

2.3.7.1 Metapopulations

In general, metapopulations consist of a group of sub-populations that exist in discrete habitat patches connected through dispersal (Hanski and Gilpin 1991, Hanski 1998). These populations can experience localized extinctions and colonizations and can be more or less connected or isolated, depending on the system (Hanski and Gilpin 1991, Hanski 1998). Five of the basic metapopulation models have been described: patchy population, mainland-island model (core-satellite population), source-sink population, classic metapopulation, and non-equilibrium metapopulation (Levins 1970, Opdam 1991, Opdam et al. 1993, 1995, Pullin 2002). Some species of butterflies, ground beetles, birds, and stream fish are known to have the metapopulation structure (Harrison et al. 1988, Thomas and Harrison 1992, Hanski et al. 1994, 1995, Gotelli 1995, Hanski and Gilpin 1997). The metapopulation theory has now taken over MacArthur and Wilson’s (1967) theory of island biogeography for the application in habitat "islands" of terrestrial landscapes. The significance of metapopulation theory for conservation planning is that multiple patches spread the risk of extinction across a large area (Gotelli 1995, p. 92). This means that even if individual populations become extinct, a set of populations (i.e., metapopulation) can persist for a long time.

2.3.7.2 Effects of Isolation and Metapopulations

Isolation measured in terms of the distance from the nearest occupied patch was the primary explanatory variable of patch occupancy by White-browed Treecreeper (*Climacteris affinis*) (Radford and Bennett 2004). Previous studies found that the effect of isolation to be particularly important for species with poor dispersal ability (Verboom et al. 1991, Hinsley et al. 1995) because if the patches are separated beyond their dispersal capability, the populations become isolated (Pearson et al. 1996). This can have grave consequences for small populations that are more likely to be subjected to extinction due to stochastic events and inbreeding (Shaffer 1987). Isolated patches would have fewer immigrants than patches close to other occupied patches (Hanski and Gilpin 1991), "thus reducing the likelihood of an isolated, declining local population being rescued (Brown and Kodric-Brown 1977) or vacant patches being re-colonized following local extinction (Hanski 1994)" (Radford and Bennett 2004).

Dispersal is a critical factor in the dynamics of spatially subdivided populations, or metapopulations (Hansson 1991, Davis and Howe 1992). Since metapopulations are connected and maintained over time by dispersing immigrants, "when structural isolation exceeds functional connectivity, dispersal is disrupted and metapopulation dynamics cease to function" (Radford and Bennett 2004). Hanski (1997) warns that many populations of rare and endangered species may be “living dead,” waiting to become extinct, if they exist in isolated patches of the collapsed metapopulation where too many patches have been lost to allow for recolonization.

2.3.8 Conclusion

“Landscape connectivity is a threshold phenomenon, in which even a minimal loss of habitat near the critical threshold ($p(c)$) is likely to disconnect the landscape, and which may have consequences for population distributions” (With et al. 1997). In my study, forests are the suitable habitat for the selected forest bird species. Therefore, the critical threshold of habitat connectivity applies to the amount of forest or the percentage of forest cover in a given landscape (buffer). As has become evident in the literature review, because landscape connectivity for the selected forest birds is strongly tied to the amount of forest (forest cover abundance), the threshold of forest cover percentage has implications for the minimum percentage of forest cover necessary to maintain the populations of these forest birds in urban regions across the eastern U.S. Because even a small loss of habitat near the threshold can lead to the loss of habitat connectivity (With et al. 1997), the threshold concept is particularly relevant and important in urban regions where remaining wildlife habitat is limited (low in proportion) and is threatened by further loss and fragmentation. However, conversely, this situation provides an opportunity for strategic planning of protecting landscape functions such as providing habitat connectivity for small mammal’s and birds’ dispersal. For example, Ahern’s (1999) defensive or protective strategy can be applied here by protecting/restoring habitat. The findings from NLMs imply that even a small amount of restoration of habitat or increase in habitat near the critical threshold may lead to re-gaining habitat connectivity that has been lost (With 1997, With and King 1997).

Another implication the critical habitat threshold for landscape planning is that habitat loss may be compensated by increase in connectivity; habitat may still remain connected at a low proportion by a landscape plan or strategy that would intentionally connect remaining habitat patches. Greenways and ecological networks are examples of such planning concepts/strategies that have potential for providing much needed connection among remaining habitat.

The focus of my study is urban regions where the proportion of suitable habitat (i.e., forests) for the selected fragmentation-sensitive, forest birds tends to be low to begin with. Empirical studies (e.g., Lande 1987, 1988, Haila 1990, Haila et al. 1993, Andrén 1994) show that when the proportion of suitable habitat is low (10-30%), further loss of habitat leads to rapidly increased distances between the habitats, leading to sudden loss of connection of formally connected habitat. Because these low ranges (10-30%) of remaining wildlife habitat is expected in urban areas, conservation measures and landscape planning strategies to protect and/or restore habitat have particular importance in the urban environment. The results of my study will have implications for planning the appropriate percentage of forest cover and the configuration of forest cover for the breeding forest bird populations in urban regions across the eastern U.S. The results of the study may encourage deliberate management of forest cover to maintain connectivity for these birds. Of course, the threshold of habitat connectivity is a complex phenomenon, depending not only on the size and isolation of habitat patches, the land uses that surround the habitat, but also the movement and dispersal of organisms or particular ecological processes of interest.

Acknowledging the difficulty in identifying the critical threshold of habitat connectivity, the concept itself and some empirical evidences for its existence merit the consideration and development of proactive conservation planning and management strategies to prioritize and protect habitat before the threshold is crossed. Alternatively, the threshold value can be used as a goal to restore habitat. Protecting and enhancing habitat connectivity has a high priority in conservation planning in urbanizing landscapes where habitats are being lost, degraded, and fragmented.

2.4 Application of Landscape Ecological Theories and Principles to Landscape Planning

2.4.1 Introduction

The purpose of this section is to set a stage for my original contribution to landscape ecological planning. The literature review has allowed me to articulate five key themes that serve as a foundation for an original landscape ecological planning framework that I will develop in chapter 4. The critical themes are: threshold response (discussed in section 2.1), habitat fragmentation (discussed in section 2.2), adaptive planning (section 2.4.6.1), connectivity in landscape ecological planning (section 2.4.6.2 based on section 2.3), and multifunctional landscapes (section 2.4.6.3). From sections 2.1 to 2.3, important landscape ecological concepts and theories that are significant in the development of the landscape planning model have been identified, analyzed, and critiqued from the literature. Threshold has been discussed as a spatial phenomenon and the threshold concept is useful to understand, predict, and manage habitat fragmentation and retain or restore connectivity. In this section, first, the evolution of landscape ecological planning is discussed. Second, the importance of spatial configurations of land uses to maintain ecological processes and develop sustainable landscapes is discussed. Third, other general models and landscape planning strategies, the use of alternative scenarios in landscape planning, and an abiotic-biotic-cultural resource model are discussed. Fourth, some of the representative landscape planning frameworks, models, or methods (i.e., Steinitz 1990, Steiner 1991, 2000, Ahern 1999, and Leitão 2001) are reviewed. These landscape planning models as well as the findings from the literature review will be used to develop the original landscape ecological planning framework. Fifth, key integrating themes that inform the landscape planning model are synthesized from the literature and discussed.

2.4.2 History of Landscape Ecological Planning

In this section, the evolution of landscape ecological planning in the U.S., noting some significant influences from Europe, will be described by reviewing theoretical-methodological advancements. With regards to the development of landscape ecological planning, it is only in the last 40 years or so for ecological theories to be actively integrated into spatial planning (Leitão and Ahern 2002, Ndubisi 2002a), although there are few notable exceptions prior to 1969 as described below. Planning and design works that used ecology as a key concept are counted as examples of early landscape ecological planning and discussed below.

2.4.2.1 Before the Late 1960's

During the second half of the 19th century, Frederick Law Olmsted, Sr. (1822-1903) used ecology as a basis for design and planning; he is regarded as the founder of the profession of landscape architecture in the U.S. (Fábos 1995). His influence, however, was not limited to the discipline of landscape architecture. Olmsted significantly influenced the way cities and communities were built to incorporate the idea of nature and parks (American Planning Association 2008). He was one of the first to support the City Beautiful movement and to introduce the idea of planned suburban development to the American landscape (American Planning Association 2008). He is probably most well-known for developing the concept of linked systems of parks and parkways, represented by the Fens and the Riverway in Boston (the Boston Park System, aka. the Emerald Necklace [c. 1880's])—later became the first metropolitan park system to serve multiple purposes (e.g., recreation, preservation of the natural landscape, and management of water quality) (Ndubisi 2002a, Fábos 1985, 2004, Ahern 2004). The significance of the development of the Emerald Necklace in the history of landscape ecological planning is twofold: one is the idea of connecting a series of parks into a coherent park system instead of a single park and the other is the idea that the same planned space can serve multiple purposes. (Connectivity in the context of landscape ecological planning and multifunctional landscapes are identified as key integrating themes in landscape planning and also consist of the core of my argument for developing landscape resilience. These notions will be explored fully in sections 2.4.6.2, 2.4.6.3, and chapter 4.) The Emerald Necklace was integrated into the Metropolitan Boston Park System in the 1890's by Charles Eliot (Ahern 2004)—a fine example of a connected urban ecological infrastructure for the Boston Metropolitan Area (Fábos 1995) and landscape ecological planning at a regional scale.

In Europe, Patrick Geddes (1854-1932) and his colleagues developed a regional survey method based on Frédéric Le Play's idea of "folk, work, place" and emphasized that by understanding the relationships among them, we could understand a region where the interaction between human actions and the environment takes place (Hall 2002a, Ndubisi 2002a). This approach of explicitly integrating and considering human interactions with the landscape is one of the characteristics of landscape ecology and landscape ecological planning (Naveh and Lieberman 1984, Zonneveld 1990, Forman 1995, Leitão 2001, Ahern 2002, Steiner 2002a, Farina 2006, Wu and Hobbs 2007a). Geddes encouraged a pragmatic approach when surveying by actually walking around a region to gather information on the resources, human responses to them, and the resulting complexities of the cultural landscape (Hall 2002a, Ndubisi 2002a). Understanding a region in terms of the relationship among "folk-work-place" attributes would become an underlying principle in the theory of human ecological planning proposed by Ian McHarg some fifty years later (Ndubisi 2002a).

In *The New Exploration* (1928), Benton MacKaye (1879-1975) advocated for the use of urban open space networks and the greenbelt concept to control urban sprawl (Smith 1993b, Ahern 2004). MacKaye, independent of Geddes, developed the method of regional survey and established the field of regional planning in the U.S. (Ndubisi 1997). Similar to Geddes', his regional planning approach applied human ecology (Ndubisi 1997). MacKaye's notion of regional planning was basically same as what McHarg would articulate some 40 years later: that planning is about revealing and understanding the intrinsic nature of the land (land suitability for different land uses) and that planners should try to satisfy both society's and nature's needs. MacKaye, Clarence Stein, Henry Wright, and Lewis Mumford were the leaders of the Regional Planning Association of America.

In the 1930's and 40's, in the U.S., Benton MacKaye and Aldo Leopold (1887-1948) argued that (1) humans are part of the land and its community (i.e., soils, waters, plants, and animals) and therefore, (2) humans have moral responsibilities to the land (i.e., a land ethic) (MacKaye 1962, Leopold 1966). Leopold, in his seminal book, *A Sand County Almanac with Sketches Here and There* (1949), advocated a land ethic: that people need to be the stewards of the land of which they are part and that they have the moral and ethical responsibility to protect it and its constituents. However, the interest to protect natural resources and the concerns over the environment had fallen out of public favor since Geddes and until the 1960's (Leitão 2001). This was due to seemingly unlimited natural resources and society's primary concern on economic growth and social issues; natural resources on which the country's economy is based seemed abundant, therefore no need for protection/conservation.

During the 1950's and culminating in the state of Illinois Recreation and Open Space Plan (1960) and the state of Wisconsin Outdoor Recreation Plan (1960), Philip H. Lewis, Jr. and his associates developed and used the concept of "environmental corridors" to guide landscape planning (Lewis 1996). There are several significant findings of their natural resource pattern inventory and its application to develop the plans. First, when the spatial configurations of water, wetlands, and steep topography of 12.5 percent or greater were plotted and overlaid, there emerged linear corridor patterns.

Lewis called these linear patterns “environmental corridors.” The linear pattern was consistent in both the state of Illinois and Wisconsin. Second, this regional landscape pattern contained most of the critical physical resources of the states. Third, therefore, Lewis (1996) argues that the environmental corridors can be used as a planning guide as he states, “These patterns can guide how and where future growth can be placed to avoid destroying the essential resources that sustain life” (p. 1). Fourth, when the inventory of timber/woodlands, Class A farmlands, and aquifer recharge areas is added, the environmental corridors identify the areas that are most important to protect for biological and aesthetic diversity, and where development should be prohibited (Lewis 1996). Lewis (1996) also saw the potential of environmental corridors to provide open space, recreation, enjoyment, and environmental education (i.e., to increase public’s awareness of the landscape). This concept of linked linear corridors providing multiple functions was later applied to greenway planning.

In the Illinois Recreation and Open Space Plan (1960), the information of soil surveys was first used to identify areas that would support various recreational activities. Also, the project team found the need to identify and preserve landscape personalities, the unique sense-of-place qualities of each sub-landscape. Moreover, perceptual resources such as spatial characteristics (three-dimensional space) were found useful in providing spatially diverse experience. For example, people’s experience in using trails can be enhanced by including varying three dimensional spaces. Most of these perceptual resources also fell in the environmental corridors. In the Wisconsin Outdoor Recreation Plan (1960), the interdisciplinary team needed to identify additional natural and cultural features (beyond what would be normally identified by overlaying water, wetlands, and steep topography) in the landscape that were important to recreation and the quality of life. Most of these supplementary resources were found to occupy specific sites but as it turned out, 85 to 90 percent of these locally cherished natural and cultural resources (e.g., waterfalls, caves, old mills, and wildlife habitat) were found to lie within the environmental corridors.

The important planning concept that I can take away from Lewis's projects is that a relatively few inventories of important natural resources such as surface water, wetlands, and steep slope, when overlaid, form patterns that can guide future development/conservation efforts. Moreover, the environmental corridors contain most of the critical natural and cultural resources of the state and the region. In other words, the environmental corridors can become the "form determinant" for future development—inform the areas for potential development as well as protection of key natural and cultural resources.

Until the late 1960's, landscape planning had lacked a means to integrate ecology into planning. Ecology as a field was also based on the "old" paradigm of a uni-directional, stable climax state successional model, a closed system, and an equilibrium concept. Planning has always reflected what deemed important by society (Leitão 2001), and ecology (i.e., natural resources and the environment) was not necessarily critically important to society (i.e., the general public) at that time because of seemingly unlimited natural resources. This is one of the reasons ecology was not actively integrated into planning. Another reason is that planning itself lacked methods/approaches to integrate ecological concepts into planning except for some methods above—and even these methods were not comprehensive treatments of ecological concepts as we will see below.

Ecology and the environmental issues (e.g., soil erosion, air and water pollution) did not become the central concern of society until the late 1960's when the resources began to show a sign of deterioration and depletion. People began to notice that some of what used to be seemingly unlimited natural resources are actually finite and could get depleted. Air and water pollution became a severe problem due to the rapid industrial development and the congestion of inner cities (Mumford 1961, Fishman 1982, Hall 2002a). Although early environmental thinkers such as George Perkins Marsh, Benton MacKaye, Lewis Mumford, and Aldo Leopold had warned the fragile nature of our environment and argued for the need to take care of the land (and the land's resources and ecological communities) that humans are also part of, this keen awareness and mindset were not commonly held by the public and consequently, the environmental issues took a back seat in planning. In sum, with some exceptions, ecological theories had not been actively integrated in physical planning until the late 1960's.

The environmental crisis of the 1960's and 70's ignited the public concern for the environment. The earlier landscape suitability analysis methods (such as the Natural Resources Conservation Service method, the Angus Hills method, and the Philip Lewis method) were not adequate to answer these concerns; the methods were neither "systematic, technically, and ecologically sound" enough nor "legally defensible" (Ndubisi 2002a, p. 222). These deficiencies led to the improvement on the earlier methods, and various views of a landscape and new theories have resulted in other threads of landscape planning (see Figure 8.1 in Ndubisi 2002a, p. 222). These new developments now collectively consist of landscape ecological planning, which has many methods/approaches to deal with complex human-natural interactions.

2.4.2.2 After the Late 1960's

1969 was an important year for the history of landscape ecological planning. It was the year when Ian McHarg's masterpiece, *Design with Nature*, was published and the U.S. National Environmental Policy Act (NEPA), which required all federal agencies to "initiate and utilize ecological information in the planning and development of resource oriented projects" (National Environmental Policy Act of 1969). It was also the year when Eugene Odum's compartment model was published. (Fábos 1985, Ahern 1995) Therefore, 1969 is arguably the year of the beginning of the modern landscape ecological planning.

In his masterpiece, *Design with Nature* (1969), Ian L. McHarg (1920-2001) advocated the need for urban and regional planners to consider an environmentally conscious approach to land use and described a new method for evaluating the intrinsic suitability of the land for different land uses and for implementing the design based on the intrinsic suitability. McHarg discussed the importance of using the full potential of nature in design, which, however, necessarily comes with some limitations imposed by nature (e.g., cannot develop on fragile lands). Lewis Mumford, McHarg's mentor and friend, wrote in the book's introduction: "McHarg's emphasis is not on either design or nature by itself, but upon the preposition *with*, which implies human cooperation and biological partnership" (p. viii). McHarg (1969) argued that "he (man) must become the steward of the biosphere. To do this he must design with nature" (p. 5). Therefore, his idea incorporated Leopold's land ethic.

The McHarg method or the University of Pennsylvania method (suitability analysis) is based on the intrinsic suitability of land for different land uses. First, various biophysical and cultural components of the land are identified and examined. (The layer-cake model developed by the firm Wallace, McHarg, Roberts and Todd [WMRT] later showed these components graphically.) Then, the overlay method is used to show the areas suitable for intended land use, varying gray tones representing the intrinsic value of land for a particular land use with the darkest being the most suitable. The outcome is a suitability map for each prospective land use under consideration. Overlaying each suitability map results in a composite map which shows a gradient of potential suitabilities based on multiple parameters/factors, for specific land uses. McHarg (1969) acknowledges and seems to encourage the potential for co-existence of multiple uses in areas where different use suitability overlaps.

The Staten Island Study (produced by the firm WMRT) shows a more advanced and elaborate form of the overlay method than earlier studies. McHarg and his team were asked by the Department of Parks, the City of New York, which owns much of the land, to evaluate the intrinsic suitability of the land for conservation, recreation (i.e., passive and active), and urbanization (i.e., residential and commercial-industrial developments). The study relies on a rational method: the analysis is driven by science. The method is explicit, using clear and objective criteria. It is replicatable and "can employ the values of the community in its development" (McHarg 1969, p. 115).

The core of McHarg's argument is the proposition that "any place is the sum of historical, physical and biological processes, that there are dynamic, that they constitute social values, that each area has an intrinsic suitability for certain land uses and finally, that certain areas lend themselves to multiple coexisting land uses" (McHarg 1969, p. 104). As we can see, McHarg (1969) recognizes that in some areas, multiple land uses can coexist. The concept of intrinsic suitability is critical for ensuing optimum use of the land and enhancing the social values (McHarg 1969).

For the Staten Island Study, more than 30 factors (classified into climate, geology, physiology, hydrology, soils, vegetation, wildlife habitats, and land use) were considered. Then, key factors for each land use (e.g., conservation) were selected. These raw spatial data were interpreted and reconstituted within a value system—this is where the values of the community were incorporated into the planning process. For each appropriate factor, value gradients were constructed, using varying tones of gray. Then, all the appropriate factors for each land use were overlaid, using transparent maps with the value gradients. The composite map for conservation, for example, displayed the areas most (the darkest tone) to least intrinsically suitable for conservation.

The key in the overlay method is that the natural processes represented by factors constitute social values and that the processes indicate the areas intrinsically suitable for each of the land uses considered. The method can identify areas for "not only intrinsic single uses, but also compatible coexisting ones and areas of competition" (McHarg 1969, p. 115). Conflicts of competing, equally suitable land uses can be resolved by exclusion, multiple uses, or the decision-makers reflecting the needs of the community (McHarg 1969).

Deficiency in the method is the lack of monetary values associated with the identified, intrinsically suitable areas for each land use (e.g., conservation, recreation, and urbanization). No cost-benefit analysis is incorporated in the method, which is its major weakness. Economic information on cost (specific interventions to natural resources)-benefit (ecosystem services) ratio is often inadequate. McHarg (1969) recommends a cost-benefit analysis to be conducted when actual plans for development/conservation for the areas are made.

McHarg (1969) sees the value in complementary land uses. The overlay method allows the search for areas that can support more than one use. “The recognition that certain areas are intrinsically suitable for several land uses can be seen either as a conflict or as the opportunity to combine uses in a way that is socially desirable” (McHarg 1969, p. 115). Multifunctional landscapes can be seen as a way of achieving sustainable landscapes (Ahern 2002). The concept/proposition that the same spatial configuration of land uses can achieve multiple functions or planning objectives is an important one, and it has been applied to greenway planning and ecological infrastructure (Fábos and Ahern 1996, Fábos and Ryan 2004, 2006, Benedict and McMahon 2006). The multifunctional concept thus constitutes one of the themes of landscape ecological planning and will be further explored in the later sections.

In summary, McHarg (1969), in his seminal book, *Design with Nature*, makes the following three key arguments. First, nature provides benefits (e.g., natural water purification, climatic amelioration, atmospheric pollution dispersal)—representing values (i.e., ecosystem services)—but also hazards to people such as flooding and wild fire. Second, nature is intrinsically variable: nature is not uniform but varies as a function of geology, climate, physiology, soils, plants, and animals. Therefore, third, some areas are intrinsically suitable for certain land uses (e.g., conservation, recreation, and urban development) and other areas for other land uses. Furthermore, certain areas are not only intrinsically suitable for certain land uses (single or multiple) but also intrinsically unsuitable for certain land use (e.g., dangerous to build on a floodplain). In other words, natural processes can place constraints or limitations to human use. Therefore, when planning for certain land uses and even for an entire urban region, it is critical to understand the natural processes that are operating and both intrinsic suitability and limitations which they indicate. This is planning and designing with nature.

McHarg's overlay method has made a lasting impact on the subsequent practice of landscape ecological planning. In the McHarg method, ecology played a central role in analyzing, designing, and planning the landscape (Ndubisi 1997, Steiner 2002b). Revised and modified forms of McHarg's suitability method are still being used in today's landscape ecological planning.

When Geographic Information Systems (GIS), originated in the 1960's, became more available in the 1970's, the McHarg method was widely implemented in environmental and landscape studies (Leitão 2001, Steiner 2002a). GIS enhanced land-use planners' abilities to identify the opportunities and constraints posed by a landscape's biophysical systems (LaGro 1996), and later socioeconomic variables. GIS (and the use of computers) solved many of the technical problems inherent in the method such as limitations on the factors having varying weights, the resolution of many factors, the transformation of gray to color of equal value, and the combination of colors or grays to develop composite maps (McHarg 1969). Moreover, GIS enabled the efficient data analysis of hand overlays of the McHarg method; it increased accuracy, repeatability (the process can be replicated by anyone with the same data), and reduced subjectivity (LaGro 1996). It can be argued that GIS's overlay operation and analysis are the same procedure as McHarg's overlay method and developing composite maps based on the intrinsic suitability (Tomlin 1990, Ndubisi 2002a).

Steiner (2002a) discusses the contribution of GIS: “GIS technologies offer new ways to describe, analyze, plan, and design the complexities of human settlements. GIS emerged concurrently with new ways to see and to record the surface of the planet, such as remote-sensing technologies. Whereas GIS programs map information, remote sensing creates imagery of phenomena on the surface of Earth” (p. 5). GIS and remote sensing can reveal previously unseen connections. For example, satellite imagery can produce daily climate information for settlements. GIS can then be used to overlay climate data on land-use and land-cover maps. Therefore, “GIS and remote-sensing technologies enable us to visualize relationships” (Steiner 2002a, p. 6). The advancement of computer technology and GIS has allowed more complete and objective land-use decisions based on the intrinsic suitability. A representative example is a “parametric approach” to planning, which will be discussed next.

In the early 1970’s, Julius Gy. Fábos and his colleagues of the Metropolitan Landscape Planning Model Study (METLAND) group at the University of Massachusetts at Amherst developed a landscape planning model in light of the recognition of the damage to environmental resources by the conversion of forests and agricultural lands to growing urban land uses (i.e., the “metropolitan invasion,” the term coined by Benton MacKaye in *The New Exploration* [1928]). Until the late 1950’s, landscape planning lacked a systematic approach to landscape problems (Fábos and Caswell 1977). After the early 1960’s, advances in computer technology made possible easy manipulation of landscape data and remote sensing technology availed much more detailed information about the landscape (e.g., soils and slopes), improving the accuracy of the data but also increasing the volume of information that needs to be dealt with (Fábos and Caswell

1977). These technological advances made possible “a new and quantitative approach to landscape planning, which was especially designed to deal with the improved accuracy in landscape data” (Fábos and Caswell 1977, p. 5). Fábos and Caswell (1977) call this the “parametric approach.” The characteristics of the approach are that: (1) it achieves a more precise definition of land; (2) it avoids the subjectivity of a landscape method (or approach); (3) it allows comparison between and affords greater consistency within landscape evaluation projects; and (4) it is suited to automation and computers (Fábos and Caswell 1977). The METLAND model is a parametric landscape assessment procedural model which could demonstrate the consequences of urbanization on landscape resources.

The METLAND model provides a landscape planning framework and landscape assessment procedures. The objectives of the METLAND model are: (1) to develop a procedure which would assess and provide quantitative values for a variety of major environmental characteristics which should be considered by decision-makers; (2) to quantify composite landscape values; (3) to demonstrate the planning utility of quantifying individual and composite landscape values; and (4) to build in a flexibility to accommodate various user needs (e.g., conservationists, developers, and farmers) to an efficient and simple model (Fábos and Caswell 1977).

The METLAND model consists of three phases (assessment, evaluation, and implementation). First, assessment is conducted on four components: (1) special resource (renewable [e.g., water], non-renewable [e.g., sand and gravel], and aesthetic-cultural resources); (2) hazards (air pollution, noise pollution, and flooding); (3) development suitability (physical, topoclimatic, and visual variables); and (4) ecological suitability (i.e., ecosystem structure and function and the implications of such structure and function in land use decisions). Individual as well as composite assessment is conducted using quantifiable metrics and the resulting maps after the application of the metrics are displayed. Second, evaluation is conducted by weighting the assessment results. Public participation is included to determine landscape resource values (e.g., water quality, water supply, agricultural and wildlife productivity, and visual quality) and desired growth policies. Because the weights on these resource values differ among stakeholders (e.g., developers, conservationists, farmers), deciding on the weighting scheme is a fundamental issue. The steps of evaluation include: preparing composite assessment maps using each weighting procedure, making compromise among the different weights used, building scenarios that correspond to land use and growth objectives, and evaluating the consequences of each scenario on the compromise composite assessment of the landscape. Finally, in the implementation phase, existing planning devices/methods/techniques are identified and new devices are developed as necessary, and they are applied to implement the planning policies.

In summary, the METLAND model provides a landscape planning framework and landscape assessment procedures (together, a landscape planning model). The parametric model essentially improved on the suitability analysis with evaluation of competing landscape-allocation options (Ndubisi 2002a). The METLAND model is still a linear model but has the strength of integrating quantified environmental resources of a metropolitan region into the decision-making process along with other “values” (e.g., economic and social values) quantified according to scientific data. Although various weighting schemes represent different stakeholder values on these land resources, the model lacks the explicit inclusion of stakeholders in the planning process (for example, compare the METLAND model's planning steps/phases with those of Steiner's [1991]). The model is particularly applicable to urban fringe areas where by far the greatest land use conversions are taking place (Fábos and Caswell 1977).

In the late 1970's William Marsh introduced environmental issues into landscape planning, including the consideration and evaluation of planning alternatives (Leitão 2001, Marsh 2005). Consideration and evaluation of planning alternatives has been recognized as an important step in a planning process (Steiner 1991, Ahern 1999). Alternative scenarios produce these planning alternatives by taking different planning options/policies and then projecting them into the future (Ahern 1999, Opdam et al. 2001, Santelmann et al. 2001, 2004, Hulse et al. 2002, Nassauer et al. 2002, Steinitz et al. 2003). (The use of alternative scenarios in landscape ecological planning will be discussed more fully in section 2.4.4.3.)

2.4.2.3 Ecosystem Management and Ecosystem-based Planning

The Federal Land Policy and Management Act of 1976 laid a foundation for a commodity-based approach to all the resources (e.g., soils, forest products, drinking water, and recreation) in public lands, even later including the endangered species, managed by federal agencies (Beattie 1996, Dombeck 1996). The land was considered to be successfully managed if it produced these "commodities." The management style was reactive to specific issues and management actions lasted for a predetermined, short-time period (Beattie 1996).

Natural resource managers began to shift their approach as they realized that species richness and diversity, nutrient flow, water quality, disturbance, and resilience are best understood as a function of overall ecosystem health (Dombeck 1996, Ndubisi 2002a) and that these ecosystems are much more complex than straightforward consumer/resource relationships may suggest (Haeuber and Franklin 1996). They also realized that in addition to individual species, highly-valued ecological processes could be preserved in large ecosystems (Lee 1999). These new findings and realization led to ecosystem management and called for a more proactive approach (Dombeck 1996). Management actions, although they had a fixed time span, tended to last longer than the commodity-based approach since now resource managers needed to work over larger geographic areas and to deal with various temporal scales (Dombeck 1996, Ward 1996).

Although various federal land-use management agencies define ecosystem management in a different way, two characteristics are commonly shared: (1) management must be built on ecological science and the understanding of ecosystem functions (Christensen et al. 1996, Mangel et al. 1996, Francis et al. 2007) and (2) humans are integral components of ecosystems (Christensen et al. 1996, Mangel et al. 1996). Sustainability is ecosystem management's overall goal (Christensen et al. 1996, Mangel et al. 1996, Francis et al. 2007).

In sum, conventional management focused largely on things like resources, wildlife, and pests for a short term in a reactive manner; contemporary ecosystem management is based on the understanding of the complexity of our interactions with natural systems, and uncertainty and disturbances are understood as part of the management of ecosystems of which humans are part (Beattie 1996, Christensen et al. 1996, Dombeck 1996, Mangel et al. 1996, Fowler 2009). Ecosystem management continues in federal agencies and approaches to managing natural resources and ecosystem-based planning have integrated adaptive management (Holling 1978, Walters 1986, Lee 1993), comanagement (Berkes et al. 2003, Olsson et al. 2004, Lister 2008), and systemic management (Fowler 2009).

2.4.2.4 The 1980's Forward

The integration of landscape ecology into landscape planning has advanced since 1980's when landscape ecology, now a matured discipline in Europe, was introduced to North America (Risser et al. 1984, Forman and Godron 1986, Turner 1989), and researchers began to realize its potential for the application to planning, design, and management of land and its resources.

Since the 1980's landscape ecology has been increasingly recognized as a powerful scientific basis for land and landscape assessment, planning, management, conservation, and reclamation in North America (Naveh and Lieberman 1984, Turner 1989, 2005, Forman 1995, Leitão and Ahern 2002, Wiens and Moss 2005, Wu and Hobbs 2007a). Landscape ecology is mainly concerned with understanding spatial interactions between landscape structure and processes, and their change over time (Wiens 1976, Urban et al. 1987, Turner 1989, Forman and Godron 1986, Forman 1995, Pickett and Cadenasso 1995, Zonneveld 1995, Leitão 2001, Turner et al. 2001, Ndubisi 2002a). Zonneveld (1990) saw the use of holism that Smut had originally proposed in studying a landscape. Zonneveld (1990, 1994, 1995) argued that (1) the whole (landscape) is usually more than the sum of its parts and that (2) the landscape can and should be studied not by analyzing the composing parts separately in detail but by studying the interconnections among its composing parts. The utility of holism at an operational level, however, is a topic of much discussion (Ndubisi 2002a).

Landscape ecology has enriched planning by translating the knowledge of spatial patterns and processes into "spatial frameworks and principles for creating sustainable spatial arrangements of the landscape" (Ndubisi 2002a, p. 195). However, the development of procedures for the systematic integration of landscape ecology concepts into planning is still a major challenge (Ndubisi 2002a). Ecosystem management, conservation planning, and landscape ecological planning keep evolving and represent a more integrated approach to sustainable planning (Figure 2.7).

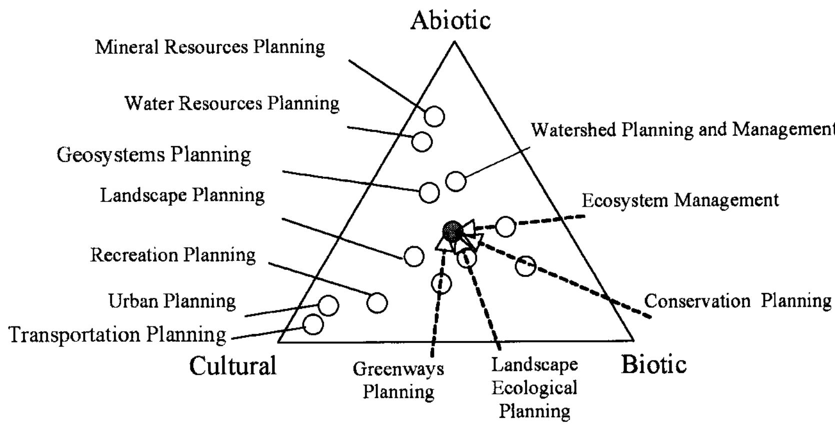

Figure 2.7: Abiotic, biotic, and cultural resource continuum and representative planning disciplines. Their locations in the triangle are relative to each other and dynamic, for several disciplines (dashed arrows) are evolving towards a more integrated perspective as represented by the center circle. It can be argued that as planning moves closer to the center of the triangle, it will be more likely to achieve sustainable objectives (*Source*: Figure 2.1 in Leitão et al. 2006, p. 28).

The advancement of GIS and remote sensing technologies has enabled detailed study and analysis of landscape structure and land use/cover. These technologies are used in landscape planning to achieve various planning objectives. Steinitz and his colleges used computer simulations to develop alternative landscape future scenarios to project the consequences of different land-use development policies/options (Steinitz and McDowell 2001, Steinitz et al. 2003), which is effectively a method of modeling landscape change

(Collinge 2009). The use of alternative landscape scenarios to evaluate the effect of various land-use policies will be discussed in more detail in section 2.4.4.3.

Joan I. Nassauer, a landscape architect who specializes in landscape ecology and design of culturally informed landscapes, has been advocating the concept of cultural sustainability: landscape patterns designed without the consideration of cultural factors are not sustainable (Nassauer 1997). A landscape embodies culture; it is a cultural expression of people's values including aesthetics. Central to Nassauer's (1995a) argument is the reciprocal relationship between landscape and culture: "culture structures landscapes and landscapes inculcate culture." Nassauer (1995a) has proposed four broad cultural principles for landscape ecology: (1) Human landscape perception, cognition, and values directly affect the landscape and are affected by the landscape; (2) Cultural conventions powerfully influence landscape pattern in both inhabited and apparently natural landscapes; (4) Cultural concepts of nature are different from scientific concepts of ecological function; and (5) The appearance of landscapes communicates cultural values. As means of achieving more precise cultural principles, Nassauer (1995a) recommends both the study of landscapes at a human scale and experimentation with possible landscapes, landscape patterns invented to accommodate ecological function. Normative scenarios (described in section 2.4.4.3) can be used to experiment with and develop possible landscapes based on these principles.

Landscape or cultural language "cues to care" is important for the long-term protection/maintenance of both remnant natural landscapes with high ecological quality and human-made landscapes with intentions to provide/restore ecological functions (Nassauer 1995b). Nassauer (1995b) argues that cultural language "cues to care," which embodies cultural values and traditions for the appearance of landscape, can be used in design to place ecological function in a recognizable context.

Nassauer (1997) argues that landscape ecology of the 21st century must be supported by cultural sustainability. “A landscape is culturally sustainable if people pay attention to its quality” (Nassauer 1997, p. 82). Her main argument is that landscapes that compel aesthetic experience are more likely to be maintained for ecological function/quality in human-dominated landscapes because the aesthetic experience can draw people to pay attention to the ecological quality of the landscape. It follows that people come to understand and appreciate the ecological function which landscape provides, and people become the stewards of the landscape. Gobster et al. (2007) also identify landscape aesthetics to be a key dimension of the relationship between natural and human systems. Nassauer’s argument applies not only to protected public lands but more so to private lands that face increasing anthropogenic disturbances.

Aesthetic experience provides explanation for why people maintain particular landscape patterns. “People make and manage landscapes not only for what they produce but for how they look and how they are *supposed* to look” (Nassauer 1997, p. 82). Therefore, Nassauer (1997) argues that policies, designs, and plans should align aesthetic expectations with ecological health. Her argument has tremendous implications for planning and designing landscapes with the intention to provide both ecological functions and aesthetic experience in human-dominated landscapes such as urban regions. Her argument can be merged with the concepts of adaptive design (Lister 2007) and designed experiments (Felson and Pickett 2005) which treat small-scale plans as experiments to test, for example, alternative spatial configurations of open space and housing and opportunities to learn from the results by monitoring key indicators of interest. Similarly, Nassauer and Opdam (2008) argue for using any intentional change of landscape pattern by design and planning as an opportunity to test landscape ecological hypotheses about the societal and environmental causes and effects of landscape patterns. Nassauer's work has contributed to establish a stronger link between the science of landscape ecology and aesthetics as a human value and cultural expression.

As human influences on the earth systems increase and become more pervasive, we need a conceptual, cultural, and ethical shift to recognize and include human activities as major components and drivers of landscape change, including ecosystems and urban regions. Interdisciplinary collaboration among ecologists, planners and designers, and social scientists, as well as transdisciplinary planning (Tress et al. 2005) are needed to develop landscapes that can sustainably provide ecosystem services while recognizably meeting societal needs and respecting societal values (Nassauer and Opdam 2008). Scenario-based planning (section 2.4.4.3) and the concepts of resilience (Holling 1973, Walker et al. 2004, Walker and Salt 2006), designed experiments (Felson and Pickett 2005), and adaptive design (Lister 2007) are useful concepts and methods to foster interdisciplinary and transdisciplinary planning and achieve more sustainable landscapes. These concepts will be further explored in the following sections and in chapter 4.

2.4.3 Importance of Spatial Configurations of Land Uses to Maintain Ecological Processes and Develop Sustainable Landscapes

In landscape ecology, it is widely accepted that the spatial arrangement of landscape elements (e.g., land uses and ecosystems) in a heterogeneous landscape has a major effect on landscape processes (Risser et al. 1984, Turner 1989, Forman 1995, Wiens 1995, Turner et al. 2001). Spatial pattern is important because it strongly controls important ecological processes such as the flow of nutrients, water, and organisms (Forman 1995). Spatial elements and their arrangement are important for landscape architects and planners, too, because these are what they manipulate to develop landscape plans (Nassauer 1997). For example, Blaschke (2006) argues that spatial arrangement matters by pointing out that providing only certain amount of certain land use types (e.g., 20% forest, 50% agricultural, 20% urban, and 10% open) is not enough to accommodate certain ecological goals such as protecting ground water recharge areas. Their arrangement and juxtaposition are also important. Hersperger (2006) also points out the importance of spatial configurations, focusing on the effects that neighboring land uses exert onto each other.

Forman (1990) asks an important question: For any landscape, or major portion of a landscape, does there exist "an optimal spatial configuration of ecosystems and land uses to maximize ecological integrity, achievement of human aspiration, or sustainability of an environment"? (p. 274). So far, no one has yet to offer an explicit and definitive answer to this question. Because landscape ecology distinguishes itself from other ecological disciplines due to its explicit attention to the spatial configuration of landscape elements for ecological processes (Turner et al. 2001), there are landscape ecology and landscape planning concepts and models that can inform an optimum spatial configuration of land uses in a region for conserving biodiversity. These models/principles include: a node(s)-and-corridor model, spatial solution, and spatial concepts.

2.4.3.1 A Node(s)-and-Corridor Model

A node-and-corridor model consists of a series of resource-rich nodes linked by corridors of access (Smith 2007). The model is proposed by biologists to explain the foraging and territory defending behavior of animals (Smith 2007). The same model can have an application to the development of greenways and ecological infrastructure. For example, the state of Maryland has been identifying and protecting the most ecologically important lands in large blocks of intact forest and wetlands, called “hubs,” linked together by linear features such as forested stream valleys, ridgelines, or other natural areas, called “corridors” (Maryland Department of Natural Resources, 2010).

A similar concept was proposed by Noss and Harris (1986) more than 20 years ago. They placed less emphasis on corridors but on a network of biodiversity hot spots. Their conceptual scheme, called a multiple-use module (MUM), consists of a protected core area or node and a buffer zone (Noss and Harris 1986). A well protected habitat core of sufficient size is needed to support interior species. Concentric buffers around the core area protect it from external influences (Baschak and Brown 1994). Different land uses can be assigned to the buffers based on their proximity to the core. Use intensity increases to the outer buffers and protection increases to the inner buffers (Noss and Harris 1986). Noss and Harris (1986) argue that the MUM network can protect and buffer important ecological entities and phenomena, while encouraging the flow of water, nutrients, organisms, and even habitat patches across space and time. They argue that the concept can work at all levels in the biological hierarchy and in all landscapes.

2.4.3.2 Spatial Solution

Forman and Collinge (1996) hypothesize that "there are spatial land-use patterns that make good ecological sense and that will conserve the bulk of nature and natural processes in any landscape or major portion thereof" (p. 538), and they call these spatial patterns collectively as a "spatial solution." The spatial solution is composed of the indispensable patterns, the aggregate-with-outliers principle, and strategic points in a landscape (Forman 1995, Forman and Collinge 1996, section 2.4.4.1). The spatial solution has emerged to address key environmental and land-use issues in any large land area (e.g., a region), specifically addressing species richness and animal movement but broadly wind, water, and soil erosion (Forman and Collinge 1996). The spatial solution is recommended as an alternative to or a supplemental to the detailed surveys of ecosystem components and functions in light of rapid human population growth, intensification of agriculture, the spread of land development, and limited resources and time to conduct detailed surveys of each ecological component (Forman and Collinge 1996). They argue that the spatial solution will be central to future land-use planning, conservation, design, management, and policy.

The conceptual spatial models such as the jaws model (Forman 1995, Forman and Collinge 1996) and the spatial solution (Forman and Collinge 1996) are powerful for their generalization but not directly applicable to a landscape plan for a real place, especially in highly heterogeneous urban/metropolitan regions with complex multidirectional dynamic processes occurring continuously (e.g., land development, land abandonment, forest loss, forest regeneration, population increase or decrease, water and species movement). For example, the aggregate-with-outliers principle (see section

2.4.4.1.2) is still a tripartite model using only three types of land use (i.e., agriculture, built, nature) and therefore has a limited capability of direct application to complex, real land-use situations. However, the aggregate-with-outliers principle as part of the patch-corridor-matrix model has been applied to Concord, Massachusetts (Ndubisi 2002a) and the Greater Barcelona Region in Spain (Forman 2008) to identify critical regional resources and also to develop/protect natural corridors. (Note that Concord is a suburban/rural community with less diversity of land use/cover types.) Landscape planners and designers need a flexible model (e.g., adaptive concept) to account for the dynamic processes occurring in urban regions. To develop such a model or a planning framework would be my original contribution to the landscape (ecological) planning field.

2.4.3.3 Spatial Concepts

Many Dutch planners argue for spatial concepts to interpret and apply basic spatial solutions to real places. Spatial concepts convey the essence of a plan or strategy in simple terms. Spatial concepts are often used in the framework of developing a landscape plan to express its overall goal or vision in the form of conceptual metaphors (van Lier 1998, Ahern 1999, Leitão et al. 2006). Examples of spatial concepts include: the Casco or Framework concept (Kerkstra and Vrijilandt 1990, van Buuren and Kerkstra 1993, van Lier 1998), greenways (Fábos and Ahern 1996, Ahern 1999), ecological networks (van Lier 1998, Opdam et al. 2006), and the neighborhood mosaic (Hersperger 2006). Spatial concepts, when implemented in landscape plans, can test landscape ecological theories and generate new knowledge (Ahern 1999). Since there are many possible spatial configurations to realize a spatial concept, these different spatial arrangements, each with its own hypothesis, can be compared and contrasted to select the best plan. Here is some discussion of the spatial concepts that are intended for direct application in landscape plans.

2.4.3.3.1 A Casco or Framework Concept: Stable Backbones and Flexible System

"Casco" is a framework concept used in landscape planning in the Netherlands. It originally refers to an architectural practice in which buildings are designed with only a main structural framework, allowing occupant modification (Ahern 1999). Originally, it is a way of classifying land uses based on the rate at which land use needs to change the spatial use/configuration of its elements (e.g., modern agriculture needs to implement new facilities, harvesting techniques, etc. vs. long-term planning of timber and water supply) (van Buuren and Kerkstra 1993).

In the Casco planning framework, parts of a landscape are designated as "high dynamic" and "low dynamic" areas. "High dynamic" areas undergo rapid changes or allow faster changes (urban development, intensive agriculture, active recreational uses) and thus, they are meant to be modified and accommodate the changing demands of people. Land modifying changes (force of water and wind operating on the landscape) take place slowly—thus, the naming—in "low dynamic" areas. "Low dynamic" areas include environmentally fragile areas (e.g., water recharge and discharge areas, flood plains, steep slopes) in need of protection. "Low dynamic" areas are designated based primarily on abiotic factors. It is spatially defined by the specific patterns created by the flows of groundwater and surface water in the landscape (i.e., the hydrological landscape structure) (van Buuren 1991, van Buuren and Kerkstra 1993). Within "low dynamic" network, there are opportunities for "high dynamic" functions and uses (Ahern 1999)—

coexisting uses of opposite nature or spatial separation (e.g., pockets of high dynamic areas within mostly low dynamic areas).

The hydrological landscape structure is used to locate the framework on which a landscape plan is based (van Buuren 1991, van Buuren and Kerkstra 1993, Ahern 1999). “This concept involves the planning of a landscape framework, a pattern of interconnected zones in which long-term, sustainable conditions for nature development and water supply are provided. This framework envelopes expanses of land in which dynamic agricultural and urban development is allowed. The location of the framework is based on the hydrological landscape structure and its (inter)relations at landscape level” (van Buuren and Kerkstra 1993, p. 230).

The Casco or Framework concept has been applied to planning to solve the conflicts within multifunctional landscapes: “how to create ecologically sound landscapes with possibilities for the development of conflicting types of land use (Kerkstra and Vrijlandt 1990)” (van Buuren and Kerkstra 1993, p. 220). “This involves (re)allocation of land-use types in a way that mutually negative impacts are prevented. At the same time the allocation of land-use types should correspond to the potentials of the natural physical structure" (van Buuren and Kerkstra 1993, p. 220).

A landscape plan based on the framework consists of a core conservation/preservation area for slow dynamics and includes areas outside of the core for fast dynamics suitable for urban development (or allow room for urban development) (van Buuren and Kerkstra 1993, Ahern 1999). In the world of continuous change, the Casco concept is relevant. It defines a durable and persistent framework that may endure changes while acknowledging that the surrounding landscape will (and should) change—

a combination of a durable frame with a dynamic context. This is the significance of the high and low dynamic notion and also the significance of the concept.

2.4.3.3.2 Ecological Networks

Ecological networks is another example of spatial concept, developed mostly by Dutch ecologists and landscape planners. The idea is catching up with other researchers in Europe and seems particularly applicable in conservation planning. Ecological networks is an European counterpart of American greenways although ecological networks seem to have more biodiversity conservation focus than greenways (Jongman 2004, Jongman et al. 2004, Jones-Walters 2007). Opdam et al. (2006) claim that ecological networks can bridge the paradox between reserve conservation (fixing nature in space and time) and development, which implies change. This is because ecological networks can change their components and spatial configurations without losing their conservation potential in the entire connected network. Ecological networks, greenways, and ecological (green) infrastructure will be reviewed through the lens of connectivity in planning and multifunctional landscapes in sections 2.4.6.2 and 2.4.6.3.

2.4.3.4 Conclusion

Three models and concepts addressing the spatial configuration of landscape elements at a broad scale are presented and discussed: the node(s)-and-corridor model, spatial solution, and spatial concepts. They acknowledge the importance of flows of water, nutrients, organisms, and information among landscape elements. The flows are influenced by the spatial configurations of these landscapes elements and this is why the spatial configuration of land uses/covers is important because it affects these critical flows to sustain important ecological processes and services.

2.4.4 Other Generalized Models and Methods

2.4.4.1 Forman's Spatial Solution

2.4.4.1.1 Indispensable Spatial Patterns

Forman (1995) identified four indispensable patterns (essential components of a landscape plan): (1) a few large patches of natural vegetation; (2) wide riparian vegetation corridors; (3) connectivity with wide corridors and/or stepping stones, allowing movement of key species, among the large patches; and (4) heterogeneous bits of nature throughout the human-dominated matrix (Figure 2.8). These spatial patterns are "indispensable" because if they are not present, the functions they support will not be provided (Forman 1995). Indispensable patterns themselves are top priority patterns for protection (Dramstad et al. 1996) and they should be "essential foundations of any land plan" (Forman 1995, p. 449), especially if the goals of the plan is to protect important ecological processes such as accommodating species movement and dispersal, conserving biodiversity, preventing soil erosion, and hydrology.

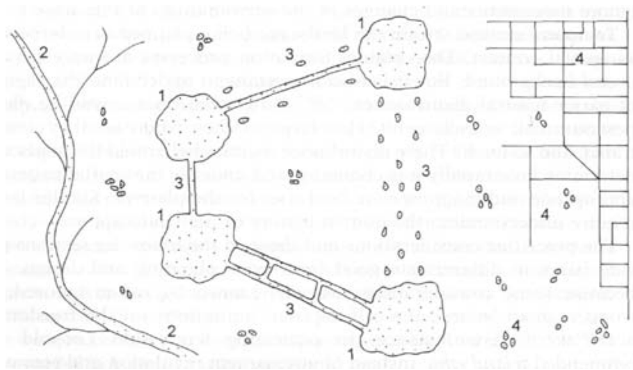

Figure 2.8: Indispensable patterns. 1 = a few large patches of natural vegetation; 2 = major stream or river corridor; 3 = connectivity with corridors and stepping stones between large patches; 4 = heterogeneous bits of nature across the matrix (*Source*: Figure 13.4 in Forman 1995, p. 452).

2.4.4.1.2 The Aggregate-with-Outliers Principle

The aggregate-with-outliers principle—being a model and a theory as well (Forman 1995)—provides a generalized spatial framework for protecting ecologically important areas and the spatial arrangement that would maximize the functioning of the three land use types (i.e., agriculture, built, nature). The principle provides a mixed land-use development framework where same land use types are aggregated, and corridors and small patches of natural vegetation are maintained throughout developed areas, with outliers of human activity (e.g., agriculture and built area) arranged along major boundaries (Figure 2.9, Forman 1995).

This is a generic model to explain and illustrate how landscape ecology principles can be applied in land-use plans. Although this principle has not been empirically tested, Forman (1995) thinks that the strengths of the principle are its flexibility in many different land use situations (e.g., suburban, agricultural, and urban) and its applicability to a range of scales.

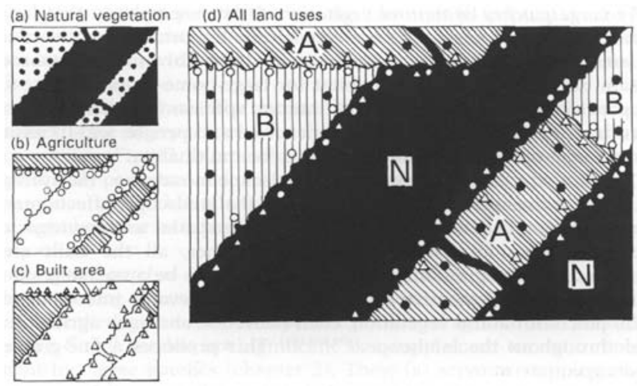

Figure 2.9: Arrangement of land uses based on aggregate-with-outliers principle. N = natural vegetation; A = agriculture; B = built area. (d) is the overlay of (a), (b), and (c) with all the symbols and hatchings (*Source*: Figure 13.1 in Forman 1995, p. 437).

The aggregate-with-outliers principle presents a way to combine the four indispensable patterns in a single landscape plan (Forman 1995). Indispensable patterns are like buildings blocks or pieces of a puzzle, and the principle shows that when they are placed together in a landscape plan, many ecological benefits such as animal movement and dispersal, risk spreading, and core habitat are provided (Turner 1989, Hansen et al. 1992, Forman 1995, Forman and Collinge 1996).

While the indispensable patterns focus more on protecting important ecological processes, the aggregate-with-outliers principle also suggests a way to maintain human activities in the target landscape. The principle actually recommends leaving "outliers" of human activity (e.g., agriculture and built area) along the boundaries of aggregated mixed land use types (see Figure 2.9). By incorporating human activities in the model, it presents a solution to the age-old dilemma of nature conservation vs. development. This makes the principle more realistic and applicable to many development situations. However, as discussed in the spatial solution (see section 2.4.3.2), the limitation of the aggregate-with-outliers principle is that it is still a tripartite model, using only three types of land use. Therefore, it has limited direct applicability to complex, real land use situations.

2.4.4.1.3 The "Jaws" Model

The jaws model (Figure 2.10) can be understood as an "explanatory" conceptual model that illustrates how large patches and connectivity can be maintained while a landscape is transformed from native vegetation to a developed condition. Thus, it is a binary, conceptual model with the purpose of explaining and showing how a binary landscape can be transformed in a manner that maintains the largest possible patch and the most connectivity for the longest time (Ahern 1999). It can be understood as a "spatial or mosaic sequence model that appears to be the ecologically optimum manner of transforming a landscape from one type to another (Forman 1995)" (Forman and Collinge 1996, p. 551).

The jaws model could arguably be a useful guide for planning but not a basis for direct application; neither can the aggregate-with-outliers principle be. The model has a potential to be used as a planning reference in landscape transformation situations (e.g., suburban landscapes and agricultural landscapes), suggesting a way to protect a remnant vegetation patch and its functions for the longest time from an ecologically less suitable land use type to which it is being converted. Because of its binary nature, the model highlights the transformation of a landscape, how the spatial configuration of the two land use types changes over time (see Figure 2.10).

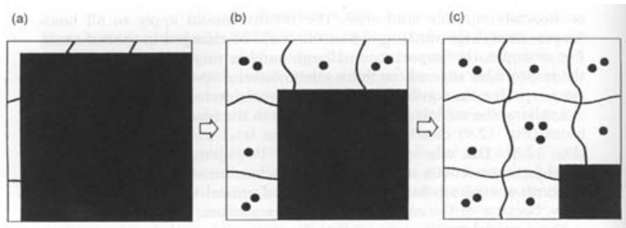

Figure 2.10: A "jaws" model of land transformation. (a), (b), and (c) are three stages showing 10%, 50%, and 90%, respectively, of the land transformed from black to white land types. Dots are small patches and curved lines are corridors (*Source*: Figure 12.11 in Forman 1995, p. 430).

2.4.4.1.4 Conclusion

Explanatory models such as the aggregate-with-outliers principle and the jaws (jaws-and-chucks) model are powerful conceptual tools but certainly not suited for directly applying to very heterogeneous urban regions with complex, multidirectional, dynamic processes occurring continuously (e.g., forest loss, forest regeneration, land development, land abandonment, population increase or decrease, water, species, etc.). Therefore, my original contribution to landscape ecological planning would be to develop a flexible model that can account for the dynamic processes occurring in urban regions. In other words, it would have means to deal with the dynamic processes.

2.4.4.2 Ahern's Four Strategies

Ahern (1995) identified and proposed four fundamental planning strategies: protective, defensive, offensive, and opportunistic. Strategy, of military origin, is "the science and art of military command exercised to meet the enemy in combat under advantageous conditions" (Merriam-Webster OnLine Dictionary 2010). Thus, strategies in planning are proactive and intended to influence the causes of problems and conflicts; strategies are more than just reacting to particular situation or achieving a specific objective (Ahern 1995). Appropriate strategies are identified and adopted according to the assessment of the landscape's current location on the trajectory of change, based on the assumption that landscapes keep changing. Ahern (2002) argues that the four planning strategies can make significant contribution to American planning theories, informed by European planning. The strength of the proposed typology of planning strategies is that it is flexible enough to be adaptable to a range of landscapes in various landscape contexts (Ahern 2002). The four principle strategies can work either individually or in combination (Ahern 1995).

Ahern's (1995) four planning strategies, taken as typology, assumes the transitional nature of landscape changes—the underlying assumption of all landscape plans that landscapes keep changing, either being modified by humans, natural successions, or disturbances. Furthermore, landscape plans themselves change the landscape. The four planning strategies are simply a typology (classification scheme), thus not meant to be a direct application to a real place.

2.4.4.3 Alternative Scenarios

The use of scenarios is another way to deal with complex and dynamic human-nature interactions to which landscape ecological planning is applied. Scenarios are "plausible stories of what might unfold in the future" (Mulvahill 2003). Scenarios should include a description of the present situation, a number of alternative futures, and the necessary steps or actions needed to link the present with the future (Nassauer and Corry 2004, Ahern 2005). Scenarios are useful in addressing the inevitable "what-if" questions: what if we use different indicator (bird) species? What if the drivers of land-use change change?—slower population growth, faster reforestation, and global climate change. In landscape planning, scenarios are used to provide alternative futures of an area (Steinitz et al. 2003). Through alternative future scenarios, spatial solutions (e.g., explicit, spatially-specific representations of land cover patterns) are developed to represent particular values, goals, or assumptions such as plan trend, maximum conservation, moderate growth, etc. The scenarios are then evaluated against the plan goals (e.g., biodiversity, water quality, accommodating population growth) and the result of the evaluation is used to decide which alternative future is most desirable, thereby informing a planning process. The development and evaluation of alternative scenarios is usually included in a planning process (e.g., Steiner's [1991] ecological planning method, Ahern's [1999] framework method).

Researchers such as Carl Steinitz, Jack Ahern, Joan Nassauer, and David Hulse have used alternative future scenarios to evaluate the consequences of changes in management scheme, resource utilization, and different paths to development/conservation (Vos and Opdam 1993, Ahern 1999, Opdam et al. 2001, Santelmann et al. 2001, 2004, Gallopín 2002, Hulse et al. 2002, Nassauer et al. 2002, Steinitz et al. 2003, Nassauer and Corry 2004). A new policy implies new future scenarios for the target landscape. Different goals (e.g., ecological, hydrological, and crop production) can be incorporated into different alternative future scenarios. Therefore, alternative future scenarios can suggest policies that could achieve specific goals or make the implications of proposed policy apparent (Nassauer et al. 2002).

Carl Steinitz and his team used different models to evaluate the scenarios for water availability, land management, and biodiversity of a portion of the Upper San Pedro River Watershed in Arizona and Sonora, Mexico, over the next twenty years

(Steinitz et al. 2003). David Hulse worked with colleagues at the U.S. Environmental Protection Agency (USEPA), the National Science Foundation, and Oregon State University on the development of spatial decision support systems for creating and evaluating alternative land and water use futures in the Willamette River Basin and elsewhere in Oregon (Hulse et al. 2002). This is an example of alternative future scenarios for the use of specific resources such as water and land uses. Using the parameters collected from empirical studies, Dutch researchers have used simulation models to evaluate the effects of different land use development/natural resource protection scenarios on the distribution and occurrence of fragmentation-sensitive forest birds and mammals (Vos and Opdam 1993).

Several researchers (Santelmann et al. 2001, Nassauer et al. 2002, Santelmann et al. 2004) described the alternative futures and their expected outcomes for agricultural watersheds in Iowa, the U.S. Nassauer and Corry (2004) advocated a normative landscape scenario approach to examine Corn Belt agricultural landscape futures under different possible federal agricultural policies. Normative landscape scenarios are different from other types of scenarios in that they depict futures that *should* be (Ahern 1999, Opdam et al. 2001, Hulse et al. 2002). Nassauer and Corry (2004) argue that normative scenarios have special potential for engaging science to build landscape policy and for exploring scientific questions in realistic simulated landscapes. They argue that the science of landscape ecology is particularly suited to develop normative scenarios by testing various land cover patterns as hypotheses as part of scenarios to provide target ecological functions that society values. This can be conducted in an adaptive planning framework (see section 2.4.6.1). They argue that normative scenarios can push scientists and policy-makers beyond their comfortable zone into developing creative landscape patterns that would accommodate ecological, economic, and cultural functions if normative scenarios are constructed with clarity, discipline, and broad interdisciplinary consultation that enables science.

In sum, a scenario includes an image of the future and a history of developments that would lead to the image (Gallopín 2002). Scenarios can be used in planning to address “what-if” questions. Scenarios are used to make conceptual models place-specific. Scenario approaches have also been suggested as a means of integrating the science of landscape ecology with landscape planning (Ahern 1999, Opdam et al. 2001). A scenario can be directly applied to a plan and scenario making can be integrated in a planning process (for example, see Ahern’s framework method). Consensus building on the preferred alternative future is an important step in the planning process. In an uncertain world, scenario planning allows a systematic assessment of the consequences of new policies with a flexibility of developing creative future visions for the area of concern (Peterson et al. 2003).

2.4.4.4 General Guidelines

Dale et al. (2000) argue that ecological knowledge of ecosystem functions can be used as a scientific basis for management and land-use decision making. The committee established by the Ecological Society of America found that five ecological principles dealing with time, species, place, disturbance, and the landscape have particular implications for land use and assuring that fundamental processes of Earth’s ecosystems are sustained (Dale et al. 2000, 2001). Based on the principles, the authors have given several guidelines (practical rules of thumb) for incorporating ecological principles into land-use decision making. The guidelines suggest that land managers should: (1) examine impacts of local decisions in a regional context, (2) plan for long-term change and unexpected events, (3) preserve rare landscape elements and associated species, (4) avoid land uses that deplete natural resources, (5) retain large contiguous or connected areas that contain critical habitats, (6) minimize the introduction and spread of nonnative species, (7) avoid or compensate for the effects of development on ecological processes, and (8) implement land-use and management practices that are compatible with the natural potential of the area (Dale et al. 2000, 2001). The authors argue that the guidelines suggest actions required to develop the science needed by land managers. These guidelines can be used as a checklist for land-use planning.

Dramstad et al. (1996) provided 55 principles of landscape ecology and showed their application to landscape design and planning to protect ecological processes and biodiversity. The authors point out the importance of spatial pattern (i.e., the arrangement of land uses and habitats) and the context of a landscape plan/design (the characteristics of the surrounding adjacent land-uses).

The guidelines for landscape planners and the landscape ecological principles for landscape architects and planners are all useful guides (so that planners and designers need to be aware of) but necessarily lack specificity to be directly applied to a specific plan/project that is unique in its institutional setting, location, the driving forces that affects, and changing over time. Ahern (2005) suggests that adaptive planning (see section 2.4.6.1) may provide a mechanism to apply these general principles and guidelines to specific locations/projects.

2.4.4.5 Abiotic-Biotic-Cultural Resource Model

Ndubisi (2002a, b) recommends the abiotic-biotic-cultural (ABC) resource survey method as a useful way of surveying and assessing resources based on abiotic (e.g., geomorphology, hydrology, physiography), biotic (e.g., flora and fauna), and cultural (e.g., stakeholder values, human use of land and changes in human activity) characteristics classified by their structural and functional attributes.

To achieve planning goals and objectives, all the resources need to be addressed in an integrated, holistic manner. The ABC resource survey method, popularized by Ndubisi (2002a, b), is a landscape ecological planning tool to identify, analyze, and evaluate important abiotic, biotic, and cultural resources in developing a landscape/land-use plan.

2.4.5 Landscape Planning Frameworks, Models, or Methods

Here, I will review several representative planning frameworks/models/methods on which I build to develop my own planning method. The methods reviewed include: Steinitz' (1990) six-level framework, Steiner's (1991) eleven interacting steps, Ahern's (1999) framework method, and Leitão's (2001) framework for sustainable landscape planning. They are all procedural methods, intended to operationalize the planning process (Ahern 2005).

2.4.5.1 Steinitz' Six-level Framework

In 1990 Steinitz proposed a six-question planning/research framework (Figure 2.11), which he later applied to his design studio projects such as the Monroe County Study and the Alternative futures of San Piedro (Steinitz 1990, Graduate School of

Design, Harvard University 2008). The framework consists of six levels (i.e., representation, process, evaluation, change, impact, and decision) and corresponding six questions that are asked at least three times during the course of a study (Figure 3.1 in Steinitz et al. 2003, p. 14). The questions are asked: (1) to define the context and scope of the work; (2) to identify the methods of study; and (3) to implement the study method—thereby creating the need to go through the six levels at least three times.

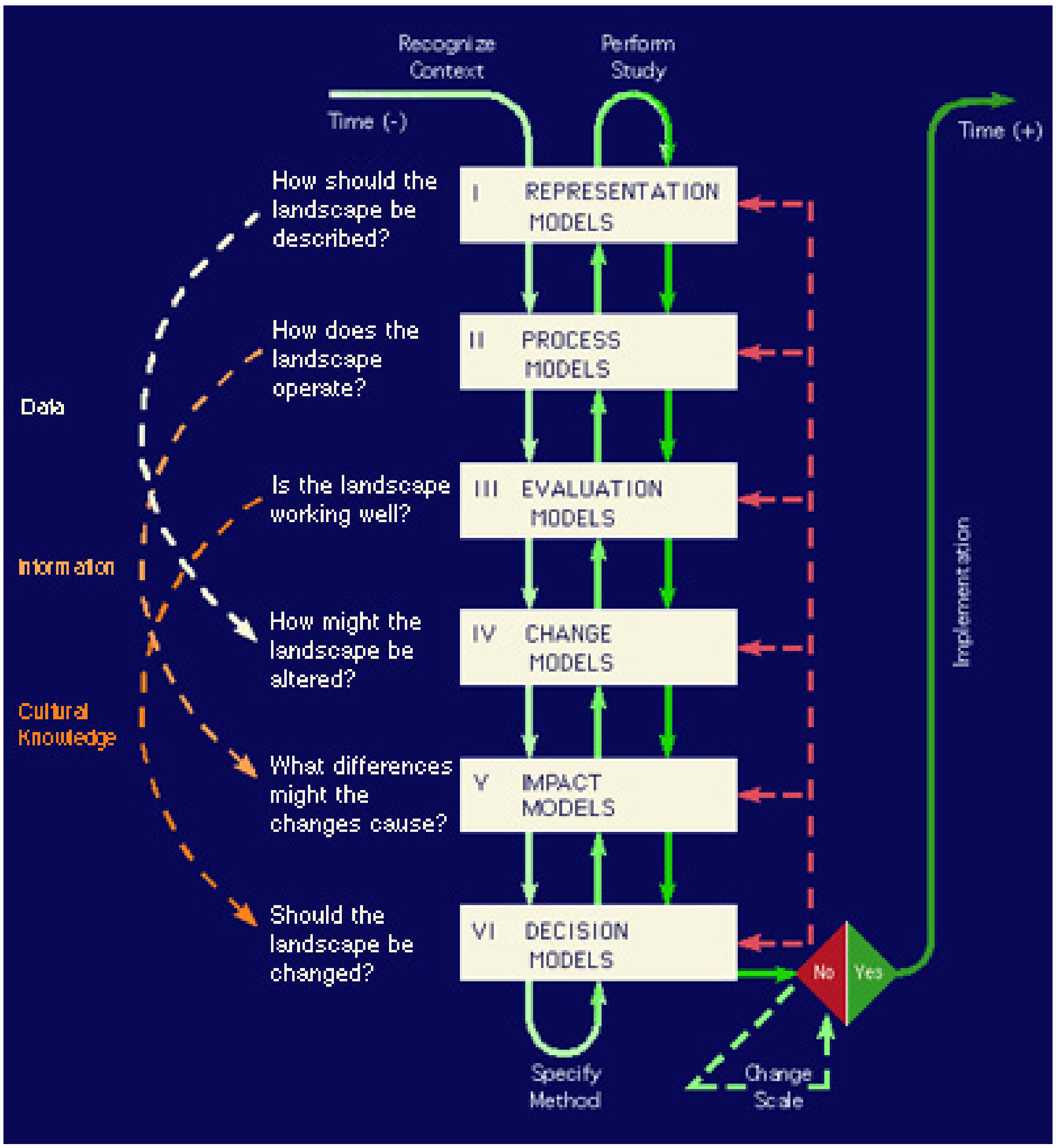

Figure 2.11: Steinitz' six-question planning/research framework (Steinitz 1990).

The unique characteristic of the framework is that it can and should be used in a reverse order to introduce "theory" (broadly defined) and to link theory more effectively with method in any project circumstance; the framework can be used as an educational tool by practitioners, academics, and students in the design/planning field (Steinitz 1990). At the first iteration, the six questions framework is used from top to bottom (see Figure 2.11). In the second time going through the six levels, the order is reversed to identify the methods of the study. Another characteristic of the model is that a feedback loop is built in at each level so that at the third iteration (implementation phase), if the decision to proceed to the next level is "No," the action is not taken, and the prior level is altered and reconsidered until the "Yes" decision is reached. (The decision to change or conserve the current landscape; not to change is a valid option after the evaluation.) The third characteristic of the framework is the inclusion of the opinions of community values. Stakeholders are responsible for making decisions. For example, when the framework was applied to develop alternative futures for the San Piedro River Basin, Steinitz et al. (2003) noted, "This study is shaped to respond to the issues and choices posed by the stakeholders. The alternative futures and the results of the assessments of their impacts are presented for stakeholder review and for the many decision processes that must precede any major action" (p. 16). The framework has been applied effectively to develop alternative futures and the evaluation of the consequences of different land and resource use policies across a range of locations (Steinitz et al. 2003, Graduate School of Design, Harvard University 2008).

2.4.5.2 Steiner's Ecological Planning Method

Steiner (1991, 2000) has developed an 11-step "Ecological Planning Method" based on McHarg's Ecological Planning Method (Ahern 2005). The method addresses multiple abiotic, biotic, and cultural goals, focusing on land-use allocation (Ahern 2005). The Ecological Planning Method is a framework for presenting information to decision-makers and for displaying "a common language, a common method among all those concerned about social equity and ecological parity" (Steiner 2000, p. 9). Therefore, the planning framework is intended to provide a common method/framework for decision-makers, citizens, planners and designers, and other professionals—a transdisciplinary planning model.

The Ecological Planning Method is composed with 11 interactive steps (Figure 2.12). The first step is problem and/or opportunity identification. The second step is the establishment of goals. The third and the fourth step involve regional-level and local-level inventory and analysis, respectively. In step 5, detailed studies such as suitability analyses are conducted to link the inventory and analysis information to the problems and goals. Step 6 is where planning concepts and options are developed. In step 7, a landscape plan is developed from these concepts. Citizen involvement and community education, although a systematic effort to involve the public occurs throughout the process, appear in step 8 where the plan is explained to the affected community members. In step 9, detailed designs are created, and the designs and plan are implemented in step 10. Finally, in step 11, the plan is administered.

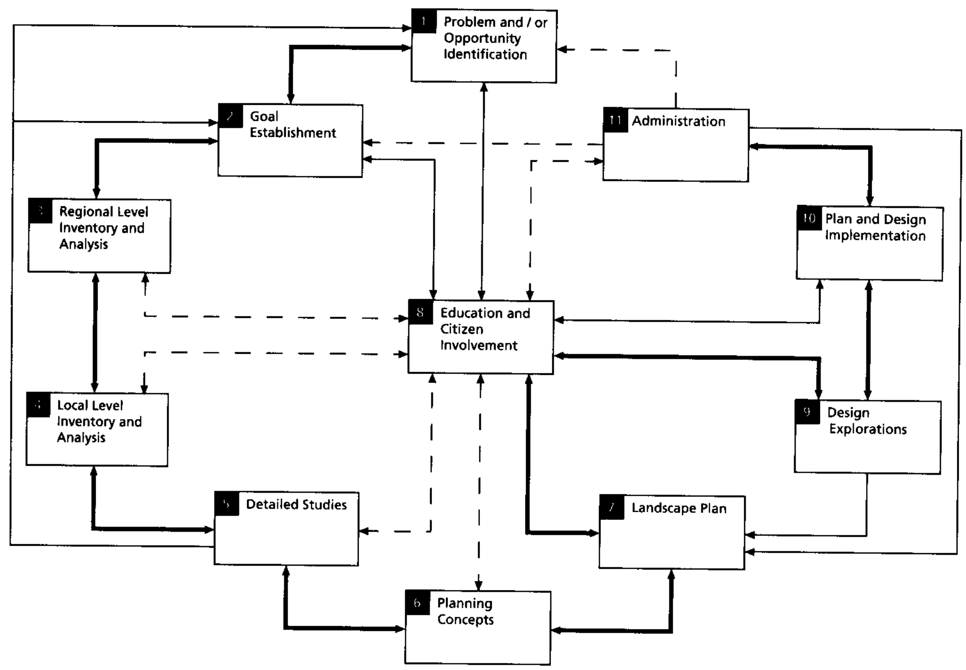

FIGURE 1.1 Ecological planning model.

Figure 2.12: Ecological planning model (*Source*: Figure 1.1 in Steiner 2000, p. 11).

Although the description of Steiner's Ecological Planning Method above may sound linear, the model is actually cyclical and very interactive. "A linear approach is inadequate for most situations—after starting to implement a plan, the original goals may change or there may be new information discovered about the environment. A feedback process is necessary to reformulate and restudy issues. In many cases, this process of review may occur repeatedly. As a result, instead of having a linear planning process, in many cases one experience a cyclic form of planning, reviewing previous stages again and again" (Steiner 2000, p. 414). The necessity and the opportunity of feedback and retroactions in order to monitor the previous results are key to an adaptive approach to planning, and will be used in my planning model as well.

The approach to planning presented by Steiner (1991, 2000) is innovative for two reasons. The first is the incorporation of ecology in planning; actually, ecology is central to the method—“the use of biophysical and sociocultural information to suggest opportunities and constraints for decision making about the use of the landscape” (Steiner 2000, pp. 9, 10). The second reason is Steiner’s stress on public participation through education and citizen involvement throughout the planning process. The importance he places on the explicit inclusion of the citizens affected by the plan in the planning process is apparent in its central placement in the diagram (Figure 2.12) and feedbacks to each planning step. The method can be applied to various strategic contexts and it employs spatial concepts in the form of design explorations at a finer scale (Ahern 2005). The Ecological Planning Method has been applied effectively to a wide range of biophysical and socio-cultural contexts (Steiner 2000).

In sum, the strength of Steiner’s Ecological Planning Method is the integration of ecology and stakeholders into the planning process. The limitations of the method are the lack of indicators for ecological sustainability (Opdam et al. 2006, Termorshuizen et al. 2007) and means of relating ecological sustainability to the interests of people and economy.

2.4.5.3 Ahern’s Planning Framework

Ahern’s framework method (Figure 2.13 below) is a way to integrate landscape ecology into landscape planning (Ahern 1999). Landscape ecology theories and concepts are applied through spatial assessments and spatial concepts (Ahern 2005). The major components of the framework are: spatial concepts, planning strategies, and future scenarios. The method is intended to explicitly address multiple abiotic-biotic-cultural resources and goals (Ahern 1995, 1999). It acknowledges potential conflicts of land use/cover patterns, and relies on spatial concepts to resolve the conflicts and seek spatial compatibility of landscape patterns (Ahern 1999, 2005). Spatial concepts are also used to design a number of future landscape scenarios, which include the means to their realization. Similar to Steinitz' method, Ahern's uses alternative future scenarios to encourage informed discussion among decision-makers, planners, scientists, and citizens on different policies and planning actions, and their potential outcomes. The discussion results are fed back to the planning process in an adaptive manner. The method takes an adaptive approach by integrating monitoring and feedback loops to adjust planning goals, strategies, and alternative scenarios to the monitoring results. It is also transdisciplinary in that interdisciplinary collaboration among scientists, planners, and stakeholders is encouraged, and the public is involved in the planning process. Although the framework method is presented linearly as seen in Figure 2.13, its intended use is nonlinear, cyclical, and iterative, and it may be initiated at any stage (Ahern 1999).

What lacks in the method is the explicit consideration of dynamics (disturbances and changes) and the means to deal with them except in scenarios. This shortcoming is common in the other models and is explained by the general lack of recognition of the importance that disturbances play in the dynamics of coupled human and natural systems at the time of these models' development.

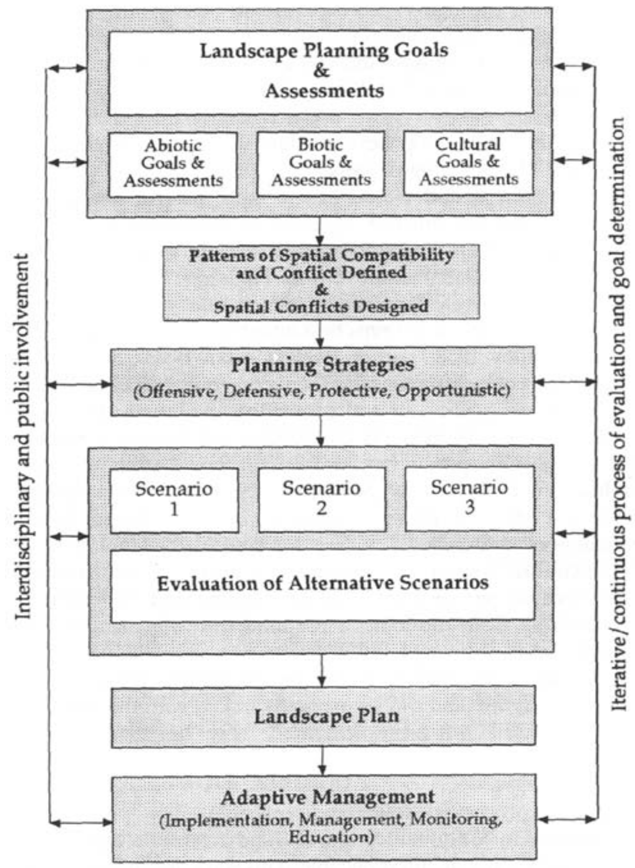

Figure 2.13: The framework method for landscape ecological planning. Even though the process is shown linearly, in actuality, it is a cyclical, continuous, participatory, and interdisciplinary process (*Source*: Figure 10.1 in Ahern 1999, p. 181).

2.4.5.4 Leitão's Framework for Sustainable Landscape Planning

Leitão (2001), Leitão and Ahern (2002), and Leitão et al. (2006) suggested a five-step planning framework for sustainable landscape planning (SLP) (Figure 2.14 below). The five planning steps (phases) are: focus, analysis, diagnosis, prognosis, and sinteresis. As clearly shown by the figure, the SLP framework is a cyclic model. The original contribution of the framework is that it showed clearly the way to use landscape metrics—suggesting 10 core set of metrics to start out—in analyzing and developing sustainable landscape plans. Landscape metrics can arguably support and be applicable to all phases of the planning process. This is an example of approaches to attempt to integrate landscape ecology into planning by the use of a suite of landscapes metrics to inform spatial planning.

Landscape metrics can help to quantify what spatial concepts try to achieve by producing numbers corresponding to each aspect of landscape structure (composition and configuration) of a proposed plan. Since ecological functions are often difficult to measure directly, landscape metrics assumed to be their surrogates are a useful tool to measure and compare the spatial configurations of alternative plans (futures) (Leitão et al. 2006). Used in combination, landscape metrics can help compare different plans or alternative scenarios for their effectiveness in achieving the goals of the plans (Leitão et al. 2006).

The major components of the SLP framework are: landscape temporal dynamics, alternative future scenarios, and landscape metrics. The inclusion of these components is hoped to bring a better understanding of the pattern-process relationships and more transparency and objectivity to the planning process (Appendix 2 in Leitão 2001). The first two components are included as tools to integrate the cultural component in the context of the SLP framework (Appendix 2 in Leitão 2001). As the name suggests, the SLP framework attempts to address not only the ecological (abiotic and biotic) component of sustainability but also the cultural component by considering landscape temporal dynamics and alternative future scenarios.

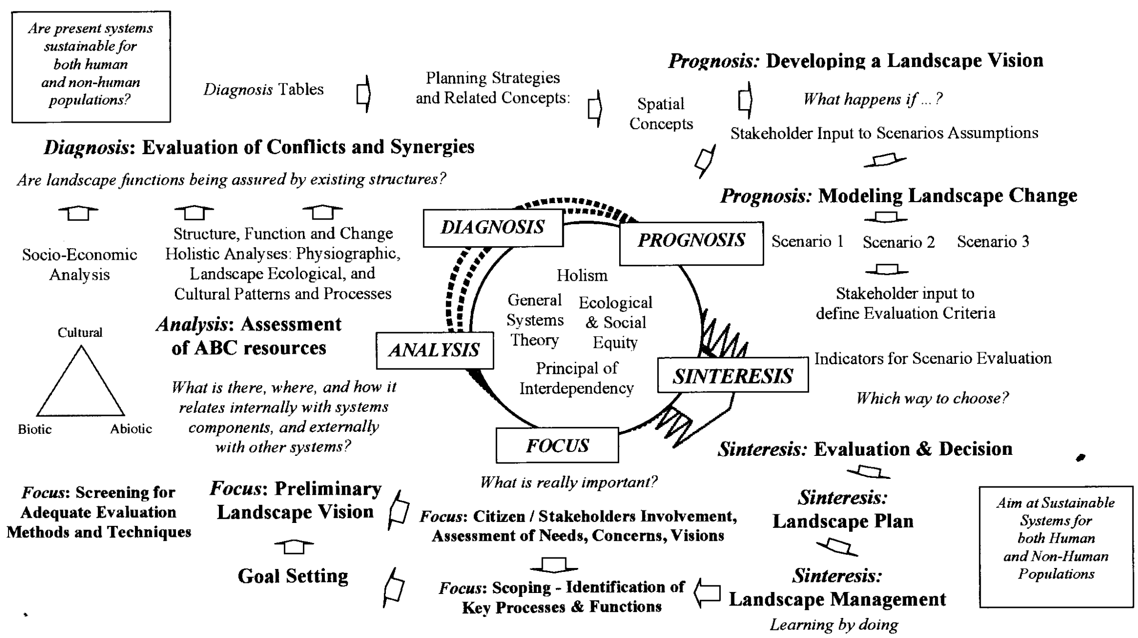

Figure 2.14: Sustainable land planning framework (*Source*: Figure 2.2 in Leitão et al. 2006, p. 44).

The major assumption of the SLP framework is that landscape metrics can represent ecological processes reasonably well. Landscape metrics do represent various landscape composition and configuration characteristics (O'Neill et al. 1988, McGarigal and Marks 1995) but the degree to which they represent actual ecological processes is still in debate (Jones et al. 2001, Wu and Hobbs 2007b, Saura and Pascual-Hortal 2007). For example, Corry and Nassauer (2005) found that landscapes metrics are not a good measure of small mammal habitat in highly fragmented landscapes, where the target patch type exists as small, narrow strips. Landscape metrics are certainly a powerful tool when a suite of metrics are used in combination and used to compare various alternative plans (Leitão et al. 2006).

The strength of the SLP framework is its attempt to include both the ecological and cultural dimensions of sustainability. In other words, it is a comprehensive model.

However, the pursuit of comprehensiveness comes at the cost of having a too busy to read diagram (Figure 2.14). The details are necessary to represent the complexity of sustainability in the planning process. However, the representation suffers from the very complexity. It takes time for a reader to follow each “branch” of the planning process and also grasp the big picture. Also, the framework claims to include a temporal aspect but this is never clear how exactly it is included in the planning process except for the use of scenarios to project some decades into the future.

2.4.5.5 Conclusion

Many landscape researchers have proposed a planning process with explicit steps, or stages (e.g., Steinitz 1990, Steiner 1991, 2000, Ahern 1999, Leitão 2001, Ndubisi 2002b). These planning frameworks or models, in general, share the following steps, or levels: assessment of existing conditions, articulation of goals and objectives, consideration of alternatives, the decision making process, and the development of a plan.

All the models/frameworks reviewed here are cyclical, iterative, and interactive, involving policy-makers, planners, scientists, and citizens in the planning process; they are adaptive and transdisciplinary models. Feedback loops are important features in all the models, incorporating and adapting to the new findings; monitoring and discussion results are fed back to the planning process. Therefore, model/scenario parameters and even plan goals can be adapted to the lessons learned—“learning by doing” (see section 2.4.6.1 for adaptive planning). There is a continuous generation of new knowledge and education in the planning process. All the models/frameworks are applicable across a range of strategic planning and abiotic-biotic-cultural contexts, as they should be because of their generalized nature. All models can be considered transdisciplinary with their inclusion of the public in the planning process and meaningful collaboration among policy-makers, planners, scientists, and citizens.

All the models include stakeholder participation to a varying degree. The inclusion of citizens and stakeholders in the planning process and decision making is shown strongly in Steiner's and Steinitz' models. Steinitz' framework includes the opinions of community values. Stakeholders are responsible for making decisions (e.g., choosing among several alternative futures). Steiner's planning framework has an explicit step of citizen involvement although this happens throughout the planning process.

Steinitz', Steiner's, and Ahern's framework share a common characteristic of being iterative at each planning step/level. In other words, a feedback loop is in built in at each step/level. Ahern's and Leitão's models are more explicit in the use of alternative future scenarios. Also, Steinitz' model, in practice, develops alternative future scenarios. Ahern's model integrates spatial concepts, as landscape ecology theories and principles are integrated through spatial concepts in landscape planning, and spatial concepts are used to resolve the compatibility and conflicts of landscape patterns. Steinitz' alternative future scenarios represent a form of spatial concept (Ahern 2005).

The concept of adaptive planning (Kato and Ahern 2008) is shown more strongly in Ahern's and Leitão's models. Ahern's and Leitão's models are more explicit in its intention to integrate landscape ecology theories and concepts into landscape planning. Steiner's model focuses more on the specific allocation of land uses on the landscape.

The key characteristics and components of these general and comprehensive landscape planning models—land suitability analysis based on forest cover, integration of landscape ecology into landscape planning through spatial assessments and spatial concepts, alternative future scenarios, adaptive planning (monitoring, feedback loops, education), and transdisciplinarity (integrating public and expert participation)—will be included in a proposed landscape planning “best practice” model, which is recommended to be practiced at each planning scale (see chapter 4). The nested planning scales together will form a proposed meta-model with landscape planning strategies and social-ecological processes linking each scale to one another (see chapter 4).

2.4.6 Key Integrating Themes that Inform my Landscape Planning Model

Some key integrating themes that form a basis for an original landscape ecological planning model are consolidated from the literature review and are discussed here. The critical themes are: threshold response (discussed in section 2.1), habitat fragmentation (discussed in section 2.2), adaptive planning, connectivity in landscape ecological planning (for a literature review of connectivity from a landscape ecological perspective, see section 2.3), multifunctional landscapes, the need to address and plan/design a landscape at multiple scales (mostly spatially at a neighborhood-, city-, and regional-scale, but temporal scales are important, too), and temporal dynamics (changes). I argue that these integrating themes in landscape ecological planning are the key to creating sustainable landscapes. When master plans are developed for a municipality or a region, it is a common practice to plan for 15-20 years ahead of time. Processes such as climate change and biodiversity certainly take longer to manifest than the duration of a typical plan/project; these processes need additional time to be understood. Therefore, I would argue that any planning model should anticipate changes and have means to accommodate them. For example, an adaptive planning model (Kato and Ahern 2008) incorporates monitoring to reduce uncertainty and feedback loops to learn by doing. The resilience thinking, which will also be integrated into my landscape planning model, embraces change as a normal part of a social-ecological system. Having a means to adapt to changes (e.g., climate change) in a model is a particularly useful feature when applied to growing urban regions where changes (disturbances) are more frequent, space is more limited, and land mosaic is finer and more heterogeneous—characteristics of urban landscapes. Adaptive planning is one such method as it acknowledges various uncertainties and deals with them by explicitly integrating them into testable hypotheses, monitoring the results, and adapting to new findings.

2.4.6.1 Adaptive Planning

Adaptive management has a tremendous potential to be applied in landscape planning and therefore, an adaptive approach to planning needs to be included in my landscape ecological planning framework. Adaptive management is a "management approach to embrace uncertainty and manage adaptively" (Light et al. 1995, p. 154). Although adaptive management has been widely practiced in natural resource and ecosystem management since the late 1970's (Walters and Holling 1990), it has not yet been widely integrated into or applied to landscape planning (Kato and Ahern 2008).

Every time a new plan is developed, planners face a unique situation. The inherent uniqueness in any "real-world" planning project lowers the likelihood that adequate data exists to support a scientifically-defensible decision. This is the common circumstance that defines planning and that planners must face routinely.

Also, planning is a time-sensitive activity. Landscape planners often do not have the luxury to wait for all the scientific data to accumulate to support planning decisions. Landscape planning addresses heterogeneous and dynamic landscapes—a moving target—by definition. Therefore, landscape planning tries to place itself ahead of these processes and to “steer” or influence them in a proactive, anticipatory way. It can be said that landscape planning is inherently prescriptive while science is often more descriptive. The imperative to act to meet political expectations and deadlines has hindered planning that takes a long time to initiate and implement and that requires monitoring the status/results long after the plan is complete. This is arguably the opposite of what adaptive planning requires.

Given the difficulty of addressing real, unique, place-based problems and the imperative for planners to act, seven principal reasons are proposed for the slow adoption of an adaptive approach in landscape planning: (1) fear of failure/liability, reluctance to accept uncertainty, (2) unsupportive institutional setting and complex and competing social values and interests, (3) lack of agreement on clearly stated goals, (4) lack of data, monitoring expertise, tradition, and culture, (5) lack of scientifically-based guidelines, (6) lack of successful precedents/models, and (7) transdisciplinary approach not widely understood or practiced (Kato and Ahern 2008).

Each landscape ecological plan must deal with its inherent site-specific uncertainties. Uncertainties exist in the effects of human activities on the environment (e.g., climate change) and in not knowing all the components that comprise a landscape and their mechanisms and interactions. A landscape plan needs to anticipate the type and magnitude of expected land use change, and to explicitly associate those changes with impacts and consequences on natural and cultural resources. This is where uncertainty becomes a major obstacle, unless there is an explicit method for identifying, understanding, and responding to all the “unknowns” that arise in the project.

The issue of uncertainty is considered central to the adaptive planning approach since it can affect every step of a planning process. The adaptive planning approach seeks to confront and minimize uncertainty by (re)assessing the feasibility and effectiveness of planning decisions and the risks inherent in each stage of the planning process. The following types of uncertainty regularly occur in landscape planning: geographical/spatial, temporal, process, transferability, and human input unpredictability (e.g., determining appropriate systems or populations of study, spatial-temporal scales, and geographic extent) (see Table 1 in Kato and Ahern 2008). These uncertainties can be reduced by strategies from science such as replication and pseudoreplication of data, the use of temporal and spatial analogues, developing multiple hypotheses and alternative models, consensus building, selection of common variables, and monitoring (Raiffa 1968, Holling 1978, Liska 1975, Schumm 1991, Neal 1993, Peck 1998, Beven 2000, Hunsaker et al. 2001, Yoccoz et al. 2001, Benda et al. 2002, Bocking 2002, Bogardi and Kundzewicz 2002, Center for Watershed Protection 2007). Since landscape planners, arguably, will never have all the information about the landscapes and systems they work in, uncertainties cannot be fully avoided. Thus, it is more intelligent for planners to know as much as they can about uncertainties and develop strategies and methods to address them—to "learn by doing," in a rigorous, systematic, and informed manner.

The uncertainty in landscape planning can be addressed by the following key concepts and principles of adaptive planning. First, an adaptive approach explicitly acknowledges uncertainty and may include it as a part of the hypotheses of a landscape ecological plan. In this way, the plan is reconceived as experiments that are tested in the real world. Second, modeling and monitoring can be used to reduce uncertainty by increasing scientific and professional understanding of a system. Third, interdisciplinary and transdisciplinary approaches help planners understand uncertainty through cooperation and sharing ideas among researchers, resource managers, decision-makers, stakeholders, and citizens. Fourth, uncertainties can be explicitly acknowledged to stakeholders and citizens who can be involved throughout a planning process. "Learning by doing" presupposes that something "uncertain" needs to be learned.

An adaptive, transdisciplinary planning process with the early involvement of stakeholders would help build a consensus among diverse stakeholders. Agreement among diverse stakeholders on clearly stated goals is a precondition for successful monitoring. Otherwise, there could be too many variables to be monitored to assess progress towards goals and to evaluate whether or not the plans/projects have achieved the intended goals after their completion.

Monitoring is a key in an adaptive planning process. For the adaptive planning process to be operational, an adaptive plan needs to identify and address key abiotic, biotic, and cultural resources, and identify scientifically-robust indicators and indices (e.g., fish index of biological integrity) to monitor before, during, and after plan implementation (Kato and Ahern 2008). Monitoring and analysis are performed to determine if the planning actions achieved the intended results (Langevelde 1994, Ahern 1999). The adaptive hypotheses integrating uncertainty can inform both planning and monitoring actions and interpretations. Based on the evaluation of the monitoring results, new and existing plans can be adapted to the lessons learned. The concept of "learning by doing" is the key here.

2.4.6.2 Connectivity in Landscape Ecological Planning

The second theme that is important in developing an original planning framework is connectivity in the context of landscape ecological planning. Connectivity from an ecological perspective has been discussed earlier (see section 2.3). Here I will focus on planning (and design) methods and strategies to achieve connectivity in a landscape. More specifically, landscape ecological planning methods and strategies to restore, create, and protect connectivity (e.g., greenways, ecological networks, and green infrastructure) are discussed as well as specific examples of their application in a real place or a plan.

Achieving connectivity in a landscape is important not only for the movement and dispersal of organisms but also for other ecological processes such as the transport of water and nutrients (Forman and Godron 1986, Saura and Pascual-Hortal 2007). Achieving connectivity is also important for human activities such as transportation of goods and services, movement such as commuting to work and walking to a corner grocery store, and for recreational activities such as jogging, bicycling, and rollerblading. Greenways, ecological networks, and green (ecological) infrastructure are planning concepts/methods that provide connectivity for their planning goals such as ecological protection, recreation, and historical/cultural preservation (Fábos and Ahern 1996, Ahern 2002, Fábos and Ryan 2004, 2006, Benedict and McMahon 2006). It can be argued that greenway planning, ecological networks, and green infrastructure attempt to apply the concept of connectivity to specific locations to provide or support multiple abiotic, biotic, and cultural functions. In this section, first, these planning concepts/methods are reviewed and specific examples of their application in a real place or a plan are discussed.

Second, I argue for cross-scale connectivity as a way to strengthen the connectivity. The key concept here is the connectivity of landscape elements at the same scale as well as across scales—cross-scale integration and connection. Third, three examples of the application of the concept of cross-scale connectivity to landscape ecological planning are reviewed and discussed. Finally, conclusion of the importance of connectivity in landscape ecological planning and methods to achieve connectivity is provided.

2.4.6.2.1 Greenways and Greenway Planning

Greenways and greenway planning are discussed as an example of the major application of connectivity concept (an important landscape ecology concept) to landscape planning. In other words, greenway planning is an example of landscape ecological planning that explicitly addresses connectivity issues. Greenways are networks of connected linear open spaces along natural or human-made features such as rivers, ridgelines, railroads, canals or roads. They are planned, designed and managed to connect and protect ecological, scenic, recreational and cultural/historic resources. A greenway can serve multiple purposes that are compatible with the concept of sustainable land use. (Little 1990, Ahern 1995, Erickson 2004). (Greenway goals/purposes are discussed in detail in the glossary in the appendix.) Greenway planning, a subset of landscape planning, is planning the elements and linkages that constitute greenways such as large protected areas, riparian corridors, and railroad corridors at multiple scales and for multiple purposes (Ahern 2002, Fábos and Ryan 2006). In this section, I will explain how the connectivity concept can be applied to develop greenways and how connectivity is achieved by greenways.

“Greenways are supported by theories from landscape ecology, particularly those concerning spatial configuration and connectivity” (Ahern 2002, p. 2). Connectivity is one of the three theoretical principles supporting greenways along with the co-occurrence of resources and the synergy of multiple uses (Ahern 2002). Indeed, greenways are based on linear features; by definition, they are connected. Natural corridors such as waterways, riparian corridors, and ridgelines in a landscape often form the basis for greenways because they are, by nature, connected (Smith 1993b, Fábos 1995). Greenway planning takes advantage of these naturally connected features in the landscape. These natural corridors provide various functions such as facilitating animal movement, transportation by humans, and transportation of water and nutrients as well as pollutants, disease, and pathogens; they can also act as barriers/filters for organisms, water, wind, and nutrients (Forman and Godron 1986, Forman 1995, Bennett 1998). The benefits of these functions would be lost if natural corridors were not connected. They are critical elements in a landscape and need to be protected for sustainable functioning of the landscape (Forman and Godron 1986, Fábos 1995, Forman 1995, 2008).

The co-occurrence of resources in greenways is a hypothesis that important natural and cultural resources tend to be concentrated in specific areas such as river valleys/corridors, ridgelines, and coastlines (Dawson 1995, Lewis 1996, Ahern 2002, Ribeiro and Barao 2006). The findings from Lewis’ Illinois Recreation and Open Space Plan (1960) and the Wisconsin Outdoor Recreation Plan (1960) corroborated this hypothesis. The ecological corridors—the linear pattern of co-occurrence of water, wetlands, and steep topography—contained most of the locally important natural and cultural resources (e.g., waterfalls, caves, old mills, and wildlife habitat) in need of protection (Lewis 1996). The ecological corridors identified for the plans can be considered as an early example of greenways. Another example that supports the hypothesis of co-occurrence of resources in greenways is the application of the greenway concept to the Metropolitan Area of Lisbon, Portugal, which is located along the Atlantic Ocean. When assessing the potential for greenway development, the researchers found that natural/ecological, cultural/historic, visual and recreational resources are concentrated linearly along natural features such as the Sintra range, the Atlantic coast, the natural drainage network, and the valleys (Ribeiro and Barao 2006, Machado et al. 2008). Another support of the hypothesis comes from the State of Georgia in preparation of statewide greenways. When making nominations for conservation purchase on a statewide basis irrespective of greenways, most priority conservation areas are found to be within greenway boundaries, concentrated along rivers, ridges, steep slopes, and coastal areas (Dawson 1995). If this hypothesis is valid, because the development of greenways takes advantage of the concentration of resources in a corridor form, greenways confer the connectivity advantage in a spatially-efficient manner (Ahern 2004).

Greenway planning also makes a concerted effort to connect linear open spaces, cultural features, and historic sites along human-made features such as railroads, canals, or roads into connected networks of greenway (Erickson 2004, Scudo 2006, Tan 2006). The Emerald Necklace and the Boston Metropolitan Park System are early examples of an effort to connect parks (public open space) with human-made corridors, also incorporating natural corridors (Eliot 1902, Fábos 1985, 2004). As early as 1870, Olmsted expressed his preference for connected park systems over a single large park for small social gatherings, escape from urban congestion, purification of air by trees, etc. (Olmsted 1870). Erickson (2004) observes that "[t]he cities that have achieved open space networks across metropolitan areas have been guided by strong visions about the benefits of connectivity and its contribution toward community health." For the recreational use of greenways, it is critical for pedestrian/jogging/bicycle paths/trails to be connected in order for them to function properly.

In conclusion, greenways are an example of the major application of connectivity concept to landscape planning. Greenways both make use of existing linear connected features (either/both natural and human-made) in a landscape and create connectivity by intentionally connecting these features to support specific processes and functions. The benefits of connectivity achieved by greenways are abundant—for example, mitigating/lessening the effects of habitat loss and fragmentation by protecting and connecting habitat and natural areas, providing recreation by connected parks and jogging/cycling paths, providing access to historic heritage sites and natural/scenic areas, thus facilitating environmental and cultural education, and so on. These specific process and functions otherwise may not occur if there is no connection. Greenway planning is a particularly effective strategy to protect and restore connection of important natural and cultural/historical resources in areas undergoing rapid urbanization (Ribeiro and Barao 2006). Greenways and greenway planning take advantage of the concept of connectivity as one of the theoretical basis and also support the various functions/uses that rely on connectivity.

Connectivity is achieved by structural (physical) and functional connectivity (for a more complete discussion on these terms, see section 2.3.2). It is important to provide structural connectivity for the target use/activity (e.g., continuous walking paths) but what ultimately matters is functional connectivity (e.g., birds can use “stepping stone” habitats to cross over a large area). Depending on the specific process or function, physical connectivity may not be necessary. For example, highly mobile animals such as birds can fly between physically disconnected patches as far as they are within a reasonable distance. Ultimately, connectivity depends on the particular process/use for which it is provided. Therefore, as long as the process can function or use can be conducted properly, it can be said that the connectivity for which it is intended is achieved. Connectivity can be measured by landscape configuration metrics such as nearest neighbor distance, proximity, and patch compaction (McGarigal and Marks 1995, Annex 2 in Leitão 2001). Fragmentation, the inverse of connectivity, can be inferred by patch number and patch size (Annex 2 in Leitão 2001) (For a full discussion on landscape metrics to quantify connectivity, see section 2.3.3 and chapter 3.)

2.4.6.2.2 Ecological Networks

Ecological networks could be considered as a European counterpart of North American greenways, although ecological networks’ primary concern is the protection of nature areas to ensure the long-term survival of plants and animals. Ecological networks can be defined as systems of nature reserves and their interconnections that make a fragmented natural system coherent in order to support more biological diversity than in its non-connected form (Jongman 2004). Ecological networks try to regain connection between fragmented natural and semi-natural habitats, forest and river corridors, and the ecological processes that once connected these fragmented areas (Jongman 2004). Throughout Europe, the interpretation of ecological networks varies, depending on different historical roots of nature conservation, planning, and scientific traditions, different geographical and administrative levels, different land uses, and in the end the political decision-making is dependent on actors with various land-use interests (Franco et al. 2003, Jongman et al. 2004). Here ecological networks are discussed as another application of the connectivity concept to landscape ecological planning.

An ecological network is composed of core areas, buffer zones, restoration areas, and ecological corridors (Jongman 2004, Jones-Walters 2007). Jongman (2004) discusses connectivity as a function of ecological corridors, as compared to connectedness denoting their physical structures. Ecological corridors represent links that permeate the landscape and maintain or re-establish natural connectivity by interconnecting remnants (Jongman 2004). Functional connectivity enables species movement/dispersal/migration that increases the chance of local/regional population survival (Jongman 2004). Although there may be negative ecological consequences (e.g., spread of diseases, exotic species, and disturbances, disruption of local adaptations, exposition to predators during travel, etc.) of ecological corridors (Noss 1987), because of their positive consequences, ecological corridors are valuable conservation tools for maintaining biodiversity (Beier and Noss 1998).

Ecological networks in Europe have become increasingly important in many new greenway initiatives, energized by both national-level and European Union legislation (Jongman et al. 2004). For example, nationwide environmental legislation had a strong impact on encouraging greenway planning at the regional and municipal levels in Portugal and Germany (Ribeiro and Barao 2006, von Haaren and Reich 2006).

The Dutch National Ecological Network is an example of the state-of-the-art, nationally planned conservation network. It was created in response to the crisis of the loss of biodiversity from the country ("Nature," Dutch Ministry of Agriculture, Nature and Food Quality 2008). The Dutch, living in one of the most densely populated countries in the world, have a longer, established history of integrating nature into cities compared to Americans. The National Ecological Network is one of the Dutch government's strategies to protect and restore nature as outlined in the 1990 Nature Policy Plan, published by the Ministry of Agriculture, Nature and Food Quality. The National Ecological Network aims to link and buffer large core nature areas in a coherent network spanning the entire country to ensure the survival of plants and animals (Department of Nature, Dutch Ministry of Agriculture, Nature and Food Quality 2005). The Dutch government aims to extend the network to protect 728,500 hectares of nature, which is about 20% of the total land area of the Netherlands, by 2018 (Department of Nature, Dutch Ministry of Agriculture, Nature and Food Quality 2005, Dutch Ministry of Agriculture, Nature and Food Quality 2008). In addition to nature areas, the National Ecological Network will include all national parks, wetlands, production forests and farmland, and more than 6 million hectares of water, including the Wadden Sea and the IJsselmeer ("Nature" and "Nature conservation in the Netherlands," Dutch Ministry of Agriculture, Nature and Food Quality 2008). The National Ecological Network is intended to link up with nature areas in Germany and Belgium in the future, to strengthen the Pan-European Ecological Network (Jones-Walters 2007, "Nature conservation in the Netherlands," Dutch Ministry of Agriculture, Nature and Food Quality 2008).

The systematic connection between natural areas is the key to the National Ecological Network, which is based on the notion that in order to secure the long-term future for biodiversity, it is not enough simply to establish protected nature areas (Department of Nature, Dutch Ministry of Agriculture, Nature and Food Quality 2005, von Haaren and Reich 2006, Jones-Walters 2007). To ensure the survival of populations, their habitats need to have a viable size and animals also need to be able to move freely between different (summer and winter) habitats (Department of Nature, Dutch Ministry of Agriculture, Nature and Food Quality 2005, Jones-Walters 2007). The connection between habitats gives species the freedom to disperse when circumstances in one nature area deteriorate, temporarily or structurally due to, for example, climate change (Jones-Walters 2007). The links between nature areas can also enhance the exchange of genetic material between different animal populations (Jones-Walters 2007). This is beneficial for the overall health and robustness of the species. The Dutch National Ecological Network is seen as a premier example of nationwide ecological plans where (1) the national government, working in cooperation with provincial and municipal authorities, nature conservation organizations, citizen groups, farmers and private parties, had a strong leadership role in planning and establishing ecological networks and which (2) actually implemented many of the ecological principles and concepts, especially connectivity, on the ground to support ecological functions and to protect, maintain, and enhance biodiversity (Department of Nature, Dutch Ministry of Agriculture, Nature and Food Quality 2005, Jones-Walters 2007, “Nature” and “Nature conservation in the Netherlands,” Dutch Ministry of Agriculture, Nature and Food Quality 2008, van der Windt and Swart 2008).

While many ecological networks in Europe are located in rural areas (Jongman 2002), my research focus is on urban regions. In urban areas, where a landscape mosaic tends to be finer and more heterogeneous and habitats are more fragmented, ecological networks are arguably more needed to counteract the effects of habitat fragmentation. However, the problems of short-term changes in land use, political and jurisdictional issues create a difficult environment for implementing ecological networks in urban areas (Cook 2002). Cook (2002) assessed the viability of planning an ecological network in the Phoenix, Arizona urban area by conducting structural analyses of the implemented ecological network plan. Cook (2002) found that the ecological network plan provided modest but important improvement in ecological systems in the Phoenix urban area. Zhang and Wang (2006) show that the methods which integrate landscape metrics with network analyses could facilitate the design of planning scenarios for urban ecological networks/greenways in Xiamen Island, China, a highly urbanized area as being one of China's earliest Special Economic Zones in the 1980's. Termorshuizen et al. (2007), on the other hand, argue that using the metapopulation concept as a spatially explicit ecological theory, appropriate to describe the relation between biodiversity and the pattern of ecosystem patches, ecosystem networks are useful for conserving biodiversity in intensively used regions (although intensive in agricultural use). They propose that ecological sustainability is achieved if quality, area, and configuration of the ecosystem network permit target species to persist.

Despite the criticism of the paucity of practical evidence that ecological networks work to conserve biodiversity and help animal movement (Boitani et al. 2007), the concept of ecological network has been well received in Europe with both national and

European-wide initiatives to develop ecological networks are on the way (Jongman et al. 2004, Tillmann 2005, Jones-Walters 2007).

2.4.6.2.3 Green Infrastructure

Green infrastructure, or ecological infrastructure, refers to “an interconnected green space network (including natural areas and features, public and private conservation lands, working lands with conservation values, and other protected open spaces) that is planned and managed for its natural resource values and for the associated benefits it confers to human populations” (Benedict and McMahon 2006, p. 3). Green infrastructure can support multiple ecological and social/cultural functions at multiple scales. It can be argued that green infrastructure, because of its various composing elements and interconnected networks across spatial scales, is perfectly suited to address multiple goals/uses. Green infrastructure plans apply key principles of landscape ecology to urban environments, specifically: a multi-scale approach with explicit attention to pattern-and-process relationships, and an emphasis on connectivity (Ahern and Kato 2007). This new term has been embraced by planners, designers, and others working in the environmental fields (Fábos and Ryan 2006).

Debates continue as to what green infrastructure entails. One may argue that greenways include all the aspects of green infrastructure described in Walmsley’s (2006) paper. Walmsley (2006) distinguishes green infrastructure from greenways (in the U.S.) in that green infrastructure is more ecologically focused, preserves large ecological “hubs,” and provides a framework for growth. As Walmsley notes, green infrastructure puts an emphasis on the essential quality of green space protection, rather than amenity aspects that some solely recreation focused greenway corridors may have (Fábos and Ryan 2006).

Green infrastructure is arguably an evolved form of greenways: it can include various landscape elements/features (both natural and human-made) and attempts to integrate these into a network of connected system across multiple scales (Ahern 1991, Benedict and McMahon 2006). For example, a green infrastructure can consists of individual rooftop gardens, street rain water gardens, connected to a system of neighborhood and regional park system. An adaptive management approach (Gunderson et al. 2008, Kato and Ahern 2008, Lister 2008) could be tested in a planning process for green infrastructure. The key concept is the integration of connectivity across multiple scales (Ahern 1991). Green infrastructure, with its emphasis on providing multiple functions and integrating its composing elements across scales, has a great potential as a test ground for the adaptive planning process (section 2.4.6.1) and for developing sustainable landscapes especially in urban areas.

2.4.6.2.4 Conclusion of the Review of Landscape Ecological Planning Methods that Provide Connectivity

Greenways, ecological networks, and green infrastructure are discussed as the major applications of the connectivity concept to landscape ecological planning. These planning methods can protect, restore, and create connectivity to protect important natural and cultural resources, and to assure the services/functions they can provide. For example, greenways can realize the inherent benefits of landscape connectivity by intentionally connecting open spaces along natural corridors and human-made features.

The big idea that is common to all the above-mentioned landscape planning methods is that “a system or networks of connected patches, corridors, and large areas is essential to achieve a sustainable landscape condition, by supporting essential ecological processes” (Ahern 2002, p. 130). This idea assumes that connecting patches, corridors, and large areas is a key to supporting important ecological functions (e.g., the flow of nutrients, water, species), and this is essential to achieving sustainable landscapes. This proposition or argument incorporates some key theories and principles of landscape ecology. One is the inherent benefits of landscape connectivity—supporting various ecological processes and human activities that rely on a connected system. Another is the importance of protecting the indispensable patterns (Forman 1995) for the protection of important ecological processes. Ecological corridors (Lewis 1996), found to contain most of the region’s critical natural and cultural features, provide another support for this argument that connected linear corridors can support important ecological functions by including important natural and cultural resources of a region. This is the basis for the merit of greenway planning.

The purpose of greenway planning, however, is not only about protecting ecological processes; in North America some greenways’ primary goals are to provide recreational and aesthetic functions. Protecting important cultural and historic sites is also important. The basis for the argument for multi-purpose greenways and green infrastructure is that a connected system of open space can accommodate ecological processes as well as recreational and cultural/historic functions. For example, natural corridors such as a riparian corridor and a coastline are rich in cultural and historic heritages as rivers have been used as the major means of transportation; it is natural that settlements, mills and factories, loading docks were developed near major waterways, at the confluence of rivers, and at the mouths of rivers (e.g., St. Louis, Missouri, by Mississippi River, Chicago by Lake Michigan, New York at the mouth of Hudson River, New Orleans at the mouth of Mississippi River). A river corridor, when integrated into a greenway system, can accommodate cultural and historic functions along with ecological processes.

2.4.6.2.5 Connectivity across Scales

I argue that integrating connectivity across multiple scales is the key to strengthen the connection. The key concept here, useful for planning, is that making connection stronger by not only connecting to different open space elements such as parks and river corridors at the same scale but connecting to the elements at *different* scales (e.g., street rain water gardens to local parks, and to regional greenway networks)—cross-scale integration and connection (Ahern 1991). The cross-scale connection is significant to accommodate the various scales at which ecological and cultural processes function (e.g., birds have a larger home range than amphibians) and mitigate/transmit disturbances also occurring at various scales. For example, a disturbance usually operates at a certain scale but landscape elements at a higher scale can provide a source of reorganization, “memory,” such as seed banks after disturbance (Gunderson and Holling 2002). Cross-scale connectivity relates to redundancy, response diversity, and having a buffer in face of disturbances.

Nature is full of examples of connectivity at multiple scales. For example, a river system and leaf veins show a network of connectivity at multiple scales for efficient and wide-spread transport of water and nutrients. In leaf veins, finer veins connect to thicker veins, and they connect to a few large veins; the network of veins covers an entire leaf to conduct water and nutrients essential for the survival of each cell that constitutes the leaf and the plant itself (Figure 2.15). In another example, connectivity in a nested hierarchy can be observed in a river system (e.g., first order, second order, and third order streams) (Figure 2.16). Human-made transportation systems such as U.S. road networks—local roads connecting to regionally important roads (state roads and major arteries) and to interstate highways—and the above-ground rail and subway networks in the Tokyo metropolitan region (Figure 2.17) mimic the nature's connectivity. As can be seen in these examples, the benefits of connectivity at multiple scales include: a wide and comprehensive coverage by the network; and this is an efficient coverage because of the integration of multiple scales—finer scale for a small area and coarse scale for a large area.

Figure 2.15: Connectivity in a nested hierarchy shown in leaf veins (*Source*: Joel Sartore Photography 2008).

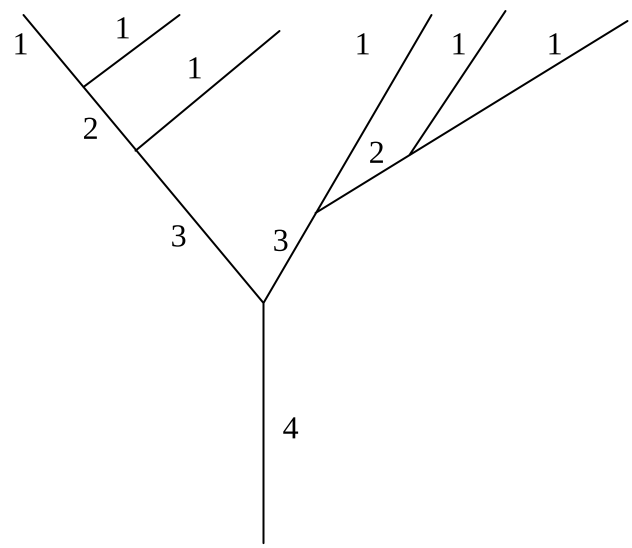

Figure 2.16: Connectivity in a nested hierarchy shown in a river system. The number represents a steam order. The higher numbered stream completely contains all the lower numbered streams.

Figure 2.17: Tokyo subway lines and some major rail lines (*Source*: JohoMaps 2006).

2.4.6.2.6 Landscape Ecological Planning Application of Connectivity across Scales

Connectivity can be strengthened by cross-scale integration and connection. Examples of the application of the concept of connectivity at multiple scales to landscape ecological planning are a conceptual framework for greenspace planning in Beijing Province, China (Li et al. 2005a), a greenspace plan for Nanjing, China (Jim and Chen

2003), and a sustainable regional plan for the Greater Vancouver region, Canada (Condon and Teed 2006).

For example, the greenspace plan of Beijing Province, China, applies the concept of connectivity across scales by connecting parks, farmlands, and forests at the neighborhood, city, and regional scale (Li et al. 2005a). The proposed plan includes ecological buffer belts at the regional scale, and green "wedges"—composed of parks, gardens, forest patches, farmlands, rivers and wetlands—and corridors at the city and neighborhood scale to control urban expansion and provide ecological services. Connectivity is the unifying theme at the three scales. For example, at the neighborhood scale, Li et al. (2005a) recommend that new parks developed be integrated into green wedges and corridors. Green wedges and corridors interact with the regional buffer belt and the large forest area to the west, and with urban parks. Patches and corridors can be linked in a network to provide connectivity among different ecosystems (Wu and Hobbs 2002).

The focus of the greenspace plan for the highly urbanized region of China is developing physical connectedness of parks, farmlands, and forests at the neighborhood, city, and regional scale. To create connectivity, Li et al. (2005a) also recommend using natural greenways such as rivers and canals; roads could be turned into greenways by incorporating roadside trees and street trees. As the authors acknowledge, the greenspace plan presented is conceptual and not elaborated in detail. There is no mentioning of specific target ecological processes for which connectivity is created except for recreation and controlling urban expansion; this is perhaps the biggest weakness of the plan.

A greenspace plan for Nanjing in China—another high-density city but more compact than Beijing—addresses the development of an integrated greenspace network also at three hierarchical scales (i.e., neighborhood, city, and the metropolitan region) (Figure 2.18, Jim and Chen 2003). The proposed comprehensive greenspace framework is intended to provide multiple functions such as guiding urban expansion, green field acquisition, recreation, wildlife habitats, and environmental benefits. It consists of green wedges, greenways and green extensions that incorporate urban green areas at the metropolitan, city, and neighborhood scale, respectively. The plan, although still at a conceptual level, has more detailed locations where these concepts will be applied and is more explicit in multifunctional aspects than that of Beijing Province (Li et al. 2005a). Although Jim and Chen (2003) did not particularly emphasize developing connectivity in the proposed plan, the result (Figure 2.18) is a clear example of the application of cross-scale connectivity. Both studies contribute to few existing studies on the application of landscape ecological planning to the high-density urban environment (Jim 2002).

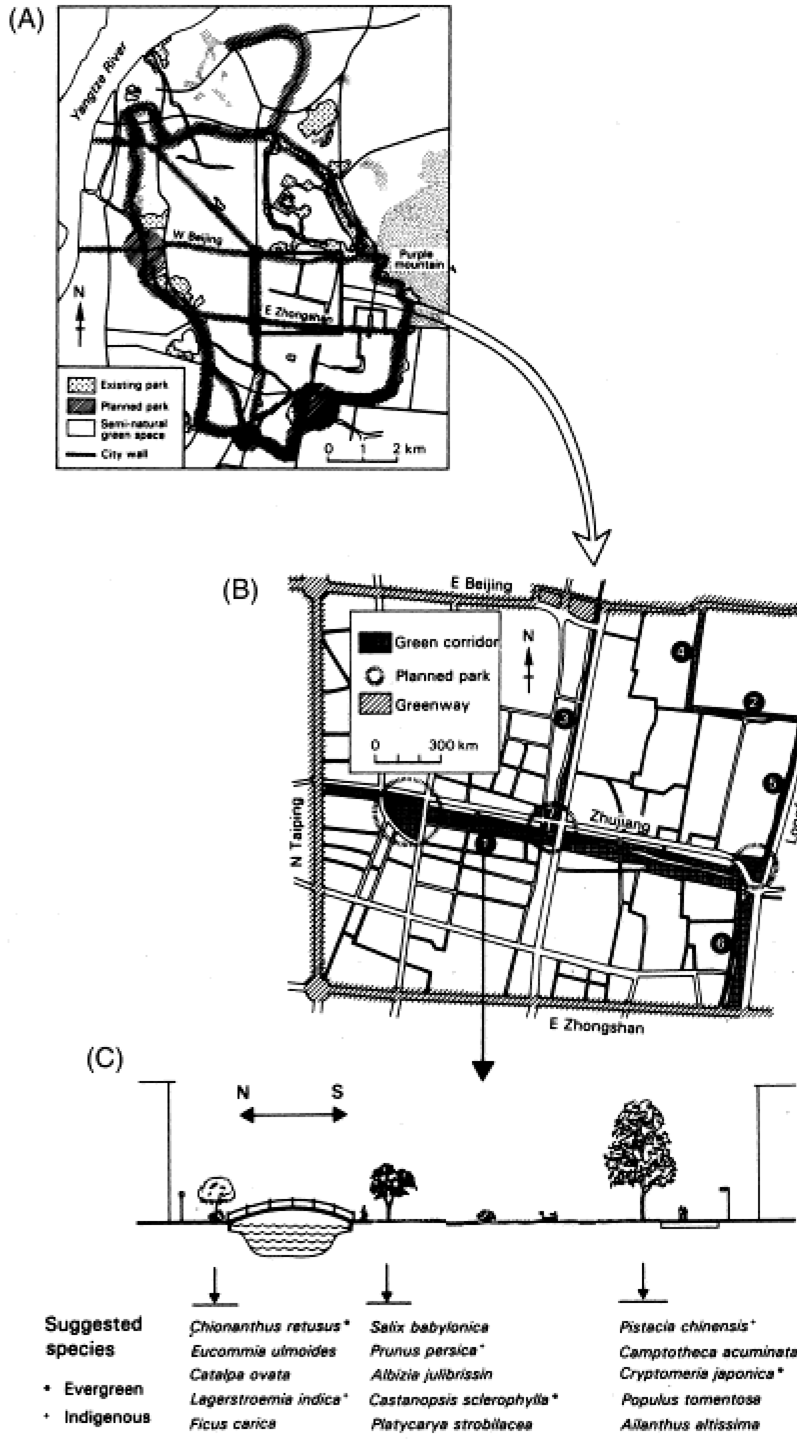

Figure 2.18: Hierarchical depiction of a proposed greenspace system including three parks and six green corridors, making use of pervasive historical canals: (A) location of the target area; (B) layout of the six green corridors (labeled 1–6); (C) landscape design and recommended tree species for the green corridors (*Source*: Figure 10 in Jim and Chen 2003).

Although not cross-scale connectivity, the sustainable vision plan of the Greater Vancouver region, Canada, employs a replication concept: “The site is to the region what the cell is to the body” (Condon and Teed 2006). The project’s operating principle is that to achieve a regional sustainability, neighborhood sustainability must be achieved, for the region is composed of the collection of neighborhoods. In the plan, connectivity is conceived as providing access to natural areas and parks, mixed use corridors accessible to all (i.e., high density commercial and residential corridors along transit routes), jobs close to home, interconnected street systems linking residents with the services they need within a five-minute walking distance, and the transformation of infrastructure networks into green and grey grids, with streets that provide natural drainage, riparian habitat, trails, and bikeways (providing multiple functions other than its original usage). All these support the principles of sustainability and explicitly relate to connectivity.

2.4.6.2.7 Conclusion

Achieving connectivity is critical for developing sustainable landscapes. By providing, protecting, maintaining, and restoring connectivity, critical ecological processes that require connectivity such as the flow of nutrients, water, and organisms can be maintained and protected. With the healthy natural resources base, human economic, social, and cultural activities—and these activities, too, require the connectivity of roads, rivers, and the Internet, for example—can flourish. The main goal of some U.S. greenways is to provide recreational opportunities for residents. Frequent contact with nature has shown to enhance the well-being of urban residents (Matsuoka and Kaplan 2008) and is also a key to fostering environmental stewardship—a sense of belonging to the larger community of the earth and everything that depends on it, and the need to take care of the plants, animals, and natural resources that are part of this community (i.e., the land ethic). The President's Commission Report (1987) supports providing access to nearby nature where people live and work. The physical connection supports the human need and preference for nearby nature and recreation (Kaplan et al. 1998). I have shown that certain recreational goals can be compatible with nature and biodiversity protection; greenways, ecological networks, and green infrastructure can achieve these multiple goals.

Connectivity is key to many important ecological functions and human socio-economic and cultural activities. The relationship between structural and functional connectivity is important in landscape planning because landscape planning manipulates structural connectivity for the purpose of protecting, creating, and restoring functional connectivity. Addressing and developing hierarchical connectivity at multiple scales (e.g., neighborhood, city, and region) is an important issue for biodiversity conservation planning because biodiversity operates at multiple scales. Using the three examples of landscape ecological plans, I have shown how this concept of connectivity across scales could be realized in a landscape for ecological and socio-cultural functions. Connectivity is an important concept in landscape ecology and landscape ecological planning can help achieve it toward developing sustainable landscapes.

2.4.6.3 Multifunctional Landscapes

A multifunctional landscape is a landscape that can support multiple planning and design objectives and values, and encompasses the concept of multiple land use, especially at a broad scale. A multifunctional landscape can be understood as a proposition that the same spatial configuration of land uses can achieve multiple functions or planning objectives, and the concept has been applied to greenway planning and ecological infrastructure (Fábos and Ahern 1996, Fábos and Ryan 2004, 2006, Benedict and McMahon 2006). An example of a multifunctional agricultural landscape may be a landscape that integrates fine-grained features such as hedgerows, patches of forest, and small wetlands into agricultural fields planted with mixed crops, which can provide multiple functions such as food production, wildlife habitat, biological pest control, water retention, etc. This fine-grained agricultural landscape is contrasted with a monoculture with a huge expanse of only one crop planted.

The spatial and temporal scale matters when discussing multifunctional landscapes. When a broad area (e.g., several kilometers) is considered, even if one land use type is assumed to provide only one function, if the area accommodates multiple land use types such as urban, open space, industrial, and commercial land use, multiple functions are provided, and therefore, it is considered to be a multifunctional landscape. In this case, the real issue which affects the functions provided is the composition and spatial configuration of land uses—the main focus of my research at a broad spatial scale. When a fine scale is considered, for example, fine-grained features in one land use area, the integration of these features may allow multiple functions. This is the example of the agricultural field above with many small features such as hedgerows, mixed crops, remnant wetlands, etc. The temporal scale matters when a land use changes over time or different uses are provided in different temporal phases in a fixed spatial extent. Thus, when the land uses in a fixed area are considered, over time, the area has a possibility to provide multiple functions even if one land use is assumed to provide only one function (this is the coarse-scale view).

The key to developing multifunctional landscapes are strategies and design and management schemes that would allow the co-existence of *competing* objectives or uses, for it is less problematic to accommodate compatible uses. The types of land-use conflicts concern temporal, spatial, and use aspects (Kato and Ahern 2009). Deciding on the strategies/schemes to address these conflicts depends on whether or not space is limited, the nature of conflicts, and the application scale (Table 2.1). The scale at which the strategies are applied is either a site (fine) or a broad, landscape scale. The site-scale strategies attempt to increase multifunctionality and deal with conflicts at a site or project scale. For example, highway overpasses can be constructed to mitigate the conflict between wildlife crossing and vehicular traffic. On the other hand, the broad-scale strategies address the entire landscape mosaic. They deal with land uses at a broad, landscape scale—for example, how to best locate a mix of competing and compatible uses across a landscape to increase multifunctionality. Various spatial concepts (e.g., greenways, the ecological network, the neighborhood mosaic concept, and the Casco concept) are included in this category. For example, greenways can support multiple uses within a connected network of linear protected areas (Ahern 2002, Erickson 2004, Fábos and Ryan 2006). While greenways and the ecological network (Opdam et al. 2006) cover only a portion of a landscape, the Casco concept (Kerkstra and Vrijilandt 1990, van Buuren and Kerkstra 1993) addresses the dynamic of the entire landscape. The concept recognizes that generally speaking, there are two types of change: one that changes quickly and the other that need stability or protection from change in order for certain ecological processes to function (for example, groundwater movement and storage). "Low dynamic" areas provide stable structure that does not change for decades and centuries. “High dynamic” areas undergo rapid changes or allow faster changes (urban development, intensive agriculture, active recreational uses) and thus, they require flexible structures that can adapt to the changes. In the context of changing landscapes (where change, whether human induced or natural cause, is the norm and cyclic), the Casco concept is significant in that it acknowledges and allows for these changes to take place in a durable framework.

Land use adjacencies are site-specific issues but their interactions shaped by the spatial configuration of land uses affect the functioning of the entire landscape. Therefore, the planning strategies to deal with adjacencies such as the neighborhood mosaic concept (Hersperger 2006) and ecological land-use complementation (ELC) (Colding 2007) affect the entire landscape. This is an example of how planning can influence the processes (e.g., ecological flows, traffic flow, species migration, seed dispersal, or nitrogen flow) through the spatial configuration of land uses. Site-specific strategies, such as creating a buffer zone and a physical barrier between conflicting uses, to deal with negative adjacencies—negative externalities (nuisances) such as noise and pollution—could be applied to the entire landscape. ELC concerns a larger structural issue, clustering different types of green spaces as compared to being isolated in the urban matrix (Colding 2007). ELC can be conceived as a way to create positive adjacencies (synergy) to support “emergent” ecological processes and greater biodiversity. These concepts/strategies dealing with the spatial configuration of landscape elements (e.g., land uses, habitat patches, ecosystems) at a broad scale are important for achieving regional planning goals such as increasing biodiversity and maintaining and enhancing ecosystem services.

Multifunctional landscapes can serve sustainable landscapes in the following ways. First, multifunctional landscapes allow the co-existence of not only compatible uses but also competing uses by the various strategies proposed (Table 2.1) and can produce advantages of synergy (Priemus et al. 2004). Second, multifunctional landscapes can (1) meet people's various demands such as recreation, industrial production, agricultural production, clean water, nature conservation, and housing, and (2) contribute to improve the quality of life (Brandt and Vejre 2004, Mander et al. 2007). This, in turn, helps develop wide constituency for these functions and a long-term support for the landscape structure that provides these functions (Ahern 1995, 2002, 2004, Rodenburg and Nijkamp 2004, Imam 2006, Tan 2006). Third, multifunctional landscapes can provide these functions efficiently: they can make an efficient use of time, space, and ABC resources by the strategies provided (Table 2.1)—this characteristic is particularly useful in urban and suburban areas, where the competition for the resources is high. For example, when ABC resources existing in a corridor form are connected by greenways, they can be conserved and utilized in a spatially-efficient manner (Ahern 2004). Fourth, the functions multifunctional landscapes provide are closely related to ecosystem services (i.e., goods and services people receive from healthy ecosystems) (Brandt and Vejre 2004). Sustaining ecological processes (e.g., the flow of water, nutrients, organisms) across a landscape and sustaining the provision of ecosystem services into the future is arguably one of the goals of sustainable landscape planning. In the landscape planning context, creating a landscape that can achieve this goal contributes to the development of sustainable landscapes. Therefore, it can be argued that creating multifunctional landscapes can help create sustainable landscapes.

In terms of the allocation of land uses across a landscape, explicit integration of human, socio-economic aspects remains to be a challenge in the evaluation of different options of allocating land use. It can be argued that a good spatial configuration of land uses is one that would optimize but not necessarily maximize certain planning objectives. Crossman and Bryan's (2009) cost-benefit approach and multi-criteria selection method (e.g., Podmaniczky et al. 2007, van der Heide et al. 2007) would aid in the decision but more research is needed in developing and testing a more integrated method (e.g., Staljanssens et al. 2003) of deciding the amount and spatial configuration of land uses across a broad landscape. To study multifunctional landscapes, inter- and transdisciplinary approaches to landscape research are required, bridging human and natural sciences (Tress et al. 2001, Boeckmann et al. 2003, Pickett et al. 2004). In the urbanizing world, multifunctionality of a landscape is a key concept to be considered in sustainable landscape planning and design.

2.4.6.4 Synthesis

An original landscape ecological planning framework will take an adaptive approach to planning, address connectivity in landscape ecological planning, and address how to develop multifunctional landscapes. Adaptive planning can provide a means to address various uncertainties involved in every step of a planning process, and planning itself needs to address a "moving target." Planners will never have complete information on a site-specific plan/project and the circumstances (the political/cultural setting where the decisions are made and the landscape itself keeps changing) surrounding the project will change while waiting for empirical data to accumulate, yet there is an imperative for planners to act. Under an adaptive approach to planning, various uncertainties (e.g., determining appropriate systems or populations of study, spatial-temporal scales, and geographic extent) can become part of adaptive hypotheses (Kato and Ahern 2008). Planning and management decisions can be re-conceived as experiments and can be implemented as adaptive plans (Ahern 2004). The results should be monitored before, during, and after the implementation, with monitoring results being fed back to adapt existing planning designs and even goals and objectives (Kato and Ahern 2008). Planners can minimize uncertainty through a monitoring program which is itself adaptive in nature, allowing them to understand the consequence of planning actions over time. However, questions remain as to what key indicators to be monitored, for how long (Ahern 2002). Planners need a planning framework that can integrate the lessons learned to the existing planning goals/objectives and therefore plans themselves (the concept of "learning by doing") and that can facilitate the continuous generation of new knowledge in a truly transdisciplinary mode, addressing abiotic, biotic, and cultural resources in a holistic, integrated way.

The inherent benefits of connectivity can be achieved by landscape ecological planning such as greenways, ecological networks, and green infrastructure. They can provide, restore, and protect connectivity, thereby protecting the functions of important natural and cultural resources. Connectivity supports various ecological and cultural processes, promoting a sustainable landscape condition (Ahern 2002).

Connectivity can be strengthened by achieving connectivity at multiple scales. The nature provides great examples of connectivity across scales: for example, leaf veins, a river system, and human blood vessels. These enable efficient and comprehensive coverage: large conduits for a coarse coverage but for a large area; fine conduits for a fine coverage but for a small area.

The idea of "collateral" uses is important for creating multifunctional landscapes. Collateral uses are the other functions/uses that are compatible (or made compatible by the strategies) with the primary objective of the plan and that can be reasonably supported by the same spatial configuration of land uses. The key to developing multifunctional landscapes is to accommodate collateral uses along with the target use/function of a plan. The benefits of multifunctional landscapes are (1) to gain spatial and economic efficiency and (2) to promote long-term cultural and political support (Ahern 2004). These attributes are also arguably necessary to develop sustainable landscapes.

Because sustainable landscapes encompass multiple dimensions (broadly, "the three Es": environment, economy, and equity [Campbell 1996]), a landscape that can serve for multiple purposes and values is a key to developing sustainable landscapes. In other words, multifunctionality is a way for a sustainable landscape to address all three dimensions (Ahern 2002). A sustainable landscape must be able to function multi-dimensionally, not for single purpose but a sustainable landscape must be able to accommodate multiple purposes and functions. Therefore, I argue that a landscape that can accommodate multiple functions/uses is more sustainable than a landscape that serves for single purpose although this kind of landscape is also necessary and may be appropriate for certain areas due to the intrinsic suitability of the land and some management restrictions/requirement.

My main argument is that because the integrating themes (i.e., adaptive planning, connectivity in landscape ecological planning, multifunctional landscapes) would help enhance/support critical ecological functions (e.g., animal movement and water and nutrient cycling) and social functions (e.g., recreation, aesthetics, and environmental education), weaving these themes into the development of a landscape planning framework would help develop sustainable landscapes. My goal is to develop an original landscape ecological planning framework which arguably can be used to develop sustainable landscapes because it can appropriately address the issues that constitute the core of sustainable landscapes from an environmental (ecological) perspective. Acknowledging the importance of the other two Es (economy and equity) and the need to simultaneously address all three Es to achieve a truly sustainable landscape, some researchers (e.g., Leitão 2001, Opdam et al. 2006) argue, and I concur, that the ecological (abiotic and biotic) component of sustainability forms the basis for addressing the other two dimensions (i.e., economy and equity) of sustainability.

Table 2.1: Types of conflicts, the strategies and examples to address the conflicts, and the spatial scale of their application.

<table>
<tr><th colspan="3">Types of Conflicts</th><th>Site Scale</th><th>Landscape Scale</th></tr>
<tr><td rowspan="9">Use Con-flicts</td><td colspan="2" rowspan="2">Time Conflicts</td><td>Park use: movement mode</td><td>Casco/Framework concept</td></tr>
<tr><td>Highway overpasses/ underpasses</td><td>Sequential phasing of an urban park development</td></tr>
<tr><td rowspan="7">Space Conflicts</td><td rowspan="2">Space Limited</td><td>Stacking (e.g., mixed use), or
Vertical separation of uses</td><td rowspan="2">By definition, NA</td></tr>
<tr><td>Create a buffer zone or a physical barrier between conflicting uses</td></tr>
<tr><td rowspan="5">Space Not Limited</td><td rowspan="5">By definition, NA</td><td>Spatial shifting for single intensive use</td></tr>
<tr><td>Spatial separation of uses (e.g., create multiple zones for different land uses, zoning)</td></tr>
<tr><td>Spatial concepts (e.g., greenways, ecological network, neighborhood mosaic)</td></tr>
<tr><td>Ecological land-use complementation (ELC)</td></tr>
<tr><td>Casco/Framework concept</td></tr>
</table>

CHAPTER 3

ROUTE-LEVEL, MULTI-SCALE ANALYSIS OF FOREST BIRD ABUNDANCE-HABITAT RELATIONSHIPS IN URBAN REGIONS ACROSS THE EASTERN UNITED STATES

3.1 Introduction

Declining biodiversity is a global concern (MA 2005, UNEP 2007) and it is attributed primarily to habitat loss, degradation, and fragmentation (Noss 1991, Tilman et al. 1994, Fahrig 1997, Peck 1998, Wilcove et al. 1998, Pullin 2002, Groom et al. 2006). Forest habitats in the suburbs are threatened by suburban sprawl and conversion to agricultural lands, which reduces forest-dependent flora and fauna, and degrades ecosystem processes and services that a healthy forest ecosystem can provide, such as water and air purification, soil erosion prevention, and carbon sequestration (Forman and Godron 1986, Rosenberg et al.1999, Marzluff 2001, Fernández-Juricic 2004). I have chosen forest birds as the focal species, species that are arguably critical to maintaining ecologically healthy conditions (Benedict and McMahon 2006). Birds have been used as the indicators of changes in habitat amount, spatial configuration (e.g., connectivity), and quality (Whitcomb et al. 1981, Morrison 1986, Bolger et al. 1997, Rosenberg et al. 1999, Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto et al. 2005, Sandström et al. 2006). Researchers have associated the distribution and abundance of birds with habitat variables (e.g., habitat composition, configuration, and quality) to create potential habitat maps of the species targeted for conservation and to determine the habitat factors that are important for the conservation of the bird species of interest (Whitcomb et al. 1981, Morrison 1986, Bolger et al. 1997). Forest birds, in particular, have been used as a response variable to measure the effect of habitat fragmentation in general due to urbanization and the conversion of forests to agricultural lands (e.g., Rosenberg et al. 1999, Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto et al. 2005, Sandström et al. 2006). These studies used forest bird species richness (i.e., the number of species) and/or the presence/absence of individual species as the indicator of the quality of urban green spaces (e.g., the composition of vegetation, the size and configuration of urban parks), or as the response variable to values of the composition and configuration of forest patches.

Some studies focused on the spatial configuration of forest patches. For example, Rosenberg et al. (1999) used Tanagers (*Piranga* spp.) and Fernández-Juricic (2004) used forest passerines as the indicator of forest fragmentation in general based on these birds' life history characteristics. Because forest-interior birds (and some ground-nesting species) are threatened by fragmentation (Marzluff 2001)—for example, susceptible to increased nest predation and brood parasitism by Brown-headed Cowbirds (*Molothrus ater*) (Robinson 1992), their abundance and occurrence can be used as the indicator of forest loss and fragmentation (Rosenberg et al. 1999, Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto et al. 2005, Sandström et al. 2006).

There are relatively few bird-habitat relationship studies in urban areas (e.g., Mörtberg and Wallentinus 2000, Fernández-Juricic 2004, Hashimoto, et al. 2005, Sandström et al. 2006) and/or at a broad spatial scale such as a regional (landscape) scale (e.g., Whitcomb et al. 1981, Askins et al. 1987, Flather and Sauer 1996, Bolger et al., 1997, Boulinier et al. 2001, Donovan and Flather 2002, Vance et al. 2003, Pidgeon et al. 2007), as compared to bird-habitat studies at the patch scale. Urban studies are few because traditionally ecologists worked in pristine environments away from human settlements (Collins et al. 2000). Regional scale studies are few because of various limitations such as time, budget, and personnel. Urban regions, or metropolitan areas, are where most people live in the United States (U.S.) (Hobbs et al. 2002) and often coincide with the areas of high biodiversity conservation priority (Groves et al. 2000, Balmford et al. 2001, Araújo 2003). The urban regional scale investigation is arguably necessary to develop a conservation plan that covers a broad area where the persistence of regional populations of birds can be ensured because: (1) some bird species (e.g., predatory species) require a large territory or a home range (Keitt et al. 1997, Thompson and McGarigal 2002); (2) some birds display metapopulation dynamics in an increasingly fragmented landscape (Opdam 1991, Opdam et al. 1995); (3) some birds, such as forest birds, have a long dispersal range and neotropical migration (Friesen et al. 1995, Robinson et al. 1995, Donovan and Flather 2002); and (4) opportunities exist to develop "smartly," lessening the impact of land use on biodiversity, mitigating the loss, and even creating new habitat. Therefore, more research is needed to investigate the bird-habitat relationship at the scale of a large urban/metropolitan region as a whole or even across multiple urban regions. Using forest-interior bird species as the indicator of broader biodiversity, an urban regional-scale study of the bird-habitat relationship would contribute to developing a regional goal for biodiversity conservation and advance landscape ecological planning that would support biodiversity in a broader urban/metropolitan region.

When considering the effects of habitat loss and fragmentation on the abundance and occurrence of forest birds in large urban regions, the critical threshold of habitat connectivity (With and Crist 1995, Wiens et al. 1997, Turner et al. 2001) is an important concept that affects the dispersal/movement of forest birds and therefore the persistence of regional forest bird populations as potential metapopulations (Opdam et al. 1995). The critical threshold of habitat connectivity is the amount (percentage) of habitat in a landscape below which the habitat becomes functionally disconnected for an organism moving across the landscape (With and Crist 1995, Fahrig 2001, Turner et al. 2001). “In landscape ecology, substantial theoretical progress has been made in understanding how critical threshold levels of habitat loss may result in sudden changes in landscape connectivity to animal movement. Empirical evidence for such thresholds in real systems, however, remains scarce” (Olden 2007). Although abrupt changes (i.e., thresholds) have been precisely defined in simulated landscapes (e.g., Gardner et al. 1987, With and King 1997, With et al. 1997, Fahrig 2001), such changes in the structure of real landscapes are not well understood. Thus, the threshold concept is an important theory to be examined in the landscape ecological data analysis, and in the context of forest birds, specifically.

Simulation models predict sudden changes in species occupancy and population persistence at the critical threshold of landscape connectivity (Gardner et al. 1987, With and Crist 1995, Fahrig 2001). This research adds to few existing empirical studies (Andrén 1994, Wiens et al. 1997, McIntyre and Wiens 1999) that tested the predictions of simulation studies by comparing multiple urban regions with different percentages of tree cover and connectivity with respect to forest bird abundance. By studying the landscape surrounding the bird survey routes in urban regions across the eastern U.S., the study expects to be able to cover a wide gradient of forest amount and spatial configuration. The study also provides a good opportunity for testing an interesting finding of earlier simulation and empirical studies that found a stronger influence of forest spatial configuration on the abundance and occurrence of forest birds when the amount of forest in the landscape is low (Cooper and Walters 2002, Flather and Bevers 2002, Betts et al. 2006b).

The main study question is: What is the relationship between forest bird abundance and the surrounding landscape characteristics, especially, forest area and its spatial configuration? The relationship will be investigated at multiple spatial scales because we often do not know *a priori* at what spatial scale the birds are responding to landscape structure characteristics (Wiens 1989, Hostetler 2001, Thompson and McGarigal 2002), in particular, the amount and spatial configuration of forest. The multi-scale analysis will be conducted by creating varying buffer distances to demarcate a “landscape” or a study corridor around bird survey routes (see 3.2.4.4).

Other study questions include:

- Do the selected birds exhibit a threshold response to the percentage of forest cover in a landscape? If so, what is the threshold percentage?
- Do important forest composition and spatial configuration factors vary when measured at different spatial scales?
- What land cover type including forest cover is the best predictor of the forest bird abundance?
- What would be a reasonable urban forest cover goal to support the selected forest birds in urban regions across the eastern U.S.?

3.2 Methods

3.2.1 Study Area

The study area is the eastern U.S.—21 states overlapping with 20 level III Eastern U.S. ecological regions or ecoregions (U.S. EPA 2007a). (Note that the southern tip of Florida and some parts of the Northeast belong to different level I ecoregions.) Ecoregions are areas that share similar ecosystems (U.S. EPA 2007a). Ecoregions are delineated by the combined abiotic and biotic characteristics such as geology, physiography, vegetation, climate, soils, land use, wildlife, and hydrology (Omernik 1987, U.S. EPA 2007a). Therefore, ecoregions can be used to divide a large area such as the conterminous U.S. into areas with similar vegetation. Since the study focus is the amount and spatial configuration of forest, the Eastern ecoregions are used to delimit the study area so that the forests in the study area share more similar abiotic and biotic characteristics at a very coarse scale than the forests in other parts of the U.S. (e.g., the Western ecoregions). The 21 states and the special district that overlap with the ecoregions are (in an alphabetical order) Alabama, Connecticut, Delaware, District of Columbia, Florida, Georgia, Kentucky, Maine, Maryland, Massachusetts, Mississippi, New Hampshire, New Jersey, New York, North Carolina, Pennsylvania, Rhode Island, South Carolina, Tennessee, Vermont, Virginia, and West Virginia (Figure 3.1). The study area contains the New York-Washington, D.C. megalopolis corridor—the most densely populated region in the U.S. The eastern U.S. is chosen as the study area because (1) the study investigates the landscape structure (composition and configuration) of forest and this area is significantly and consistently forested and (2) the types of vegetation between the East and much of the West are distinctively different, especially the arid Southwest. A study area including both the eastern and western U.S. would be too heterogeneous in terms of vegetation and would therefore complicate statistical analysis. Since ecoregions by definition include areas with similar climatic and geological features (Omernik 1987, U.S. EPA 2007a), the use of same ecoregions to delimit the study area would lessen the effects of unaccounted abiotic and biotic factors that may influence the bird-habitat relationship.

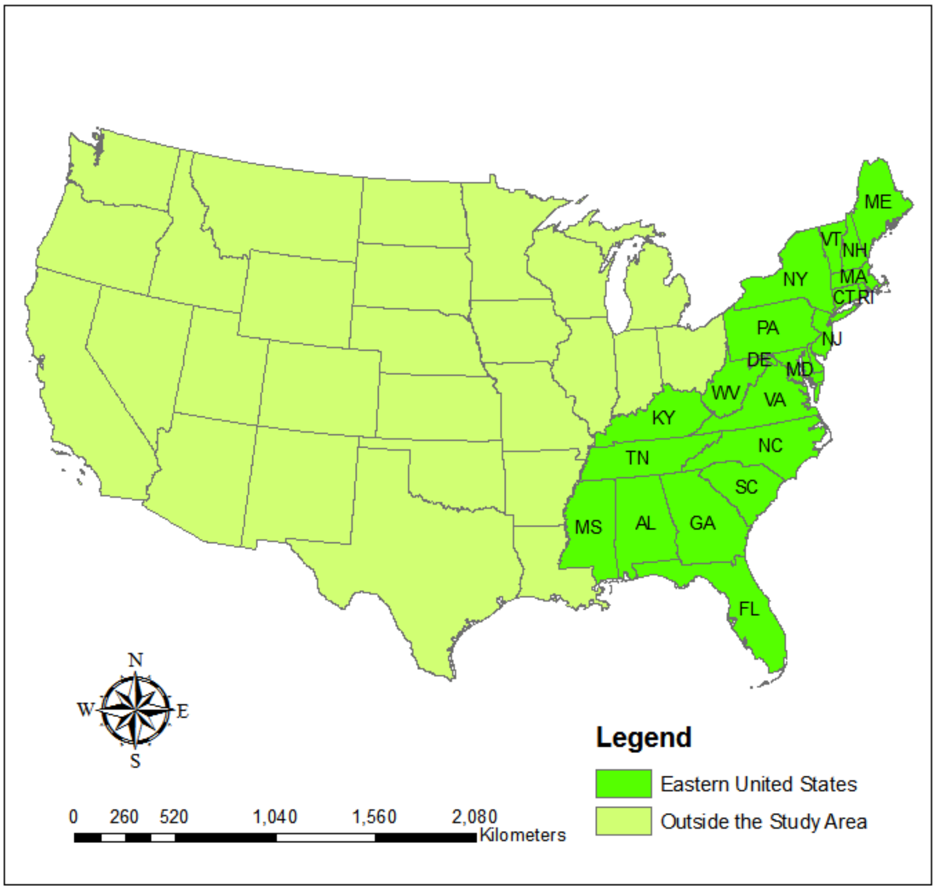

Figure 3.1: Study area in darker green: 21 eastern states and Washington, D.C.

Within the study area, the focus is on *urban regions*. The term “urban” is defined in a broad, inclusive sense rather than densely-built areas and their characteristics. The term “region” is used in the sense of regional planning practiced by Patrick Geddes and

Benton MacKaye to emphasize the importance of people's interactions with the land, including its natural and cultural resources (MacKaye 1962, Miller 1986, Ndubisi 1997, Hough 2004). According to Steiner (2002a), a region can be delineated by either biophysical or cultural characteristics, or by both in an integrating manner. Urban or metropolitan region, therefore, is defined as a spatial/geographical entity composed of interacting abiotic, biotic, and cultural resources, and can be composed of multiple jurisdictions (McDonnell and Pickett 1990, Forman 1995, 2008, Medley et al. 1995, Foresman et al. 1997, Steiner 2002a). Its boundary is determined by some measure of the intensity of urbanization, or human influences on ecosystems in the landscape (McDonnell and Pickett 1990, Forman 1995, Medley et al. 1995, Foresman et al. 1997, Steiner 2002a). The Greater Boston region or the New York metropolitan area is an example of an urban region. Forman's (2008) definition of urban region includes larger areas outside the urban core as his definition includes a metropolitan area and an urban-region ring (see Figure 1.2 in Forman 2008, p. 6).

In this study, Metropolitan Statistical Areas (MSAs) as defined by the U.S. Office of Management and Budget are used to delineate urban regions in the study area (i.e., 21 eastern states and Washington D.C.). An MSA consists of one or more core urban areas with a population of at least 50,000 and neighboring areas with strong economic and social ties to the core(s) (U.S. Office of Management and Budget 2000). To measure the ties to the urban core, the MSAs use the percentage of people from the surrounding areas (counties) commuting to the core city (or to the county that includes the core city). The MSAs use a county (or counties) as their geographic boundaries. Although in New England the power to plan usually resides in each municipality (town/city) and seldom at the level of a county, a county is likely to be a useful administrative unit for regional planning in other parts of the study area such as Maryland and Florida (Calthorpe and Fulton 2001, Steinitz and McDowell 2001, Booth et al. 2002, Weber et al. 2006). The concept of MSA—a relatively large area with a city (town) and its surrounding suburbs and perhaps even some rural areas that have economic and social connections to the core city/town—is one way to define urban areas in general. Other ways of defining urban areas, or quantifying human influences on landscapes, include: population density, housing density, percent impervious surface, ecological footprint, and most recently, the amount of CO_2 emissions (Schueler 1994, Arnold and Gibbons 1996, Brabec et al. 2002, Homer et al. 2004, Miltner et al. 2004, Solecki and Rosenzweig 2004, Purdue University 2007, U.S. Census Bureau 2007b). One big advantage of using publicly available data such as MSAs for defining urban regions is that other researchers have an easy access to the same data and thus can repeat/replicate the study. Another advantage is the low cost of acquiring the data; a researcher needs only an internet connection.

Using as large an area as a MSA to define urban regions can be problematic. Since a county is used as a unit for MSAs, some MSAs include areas that are considerably forested, such as the Berkshire region of the western Massachusetts, which can hardly be called an “urban” region (see Figure 3.2 below) in a common sense. Another limitation of using MSAs to define urban regions is that they may not include semi or peri-urban areas from adjacent counties that lack an urban core. In this broad-scale, comparative observational study, the MSAs are used to select more urbanized areas within the study area because the research interest is the planning of broad urban regions not rural areas. The study interest is to have a long gradient of forest amount and spatial distribution in urban regions. Therefore, it is acceptable to have some urban regions that are largely forested.

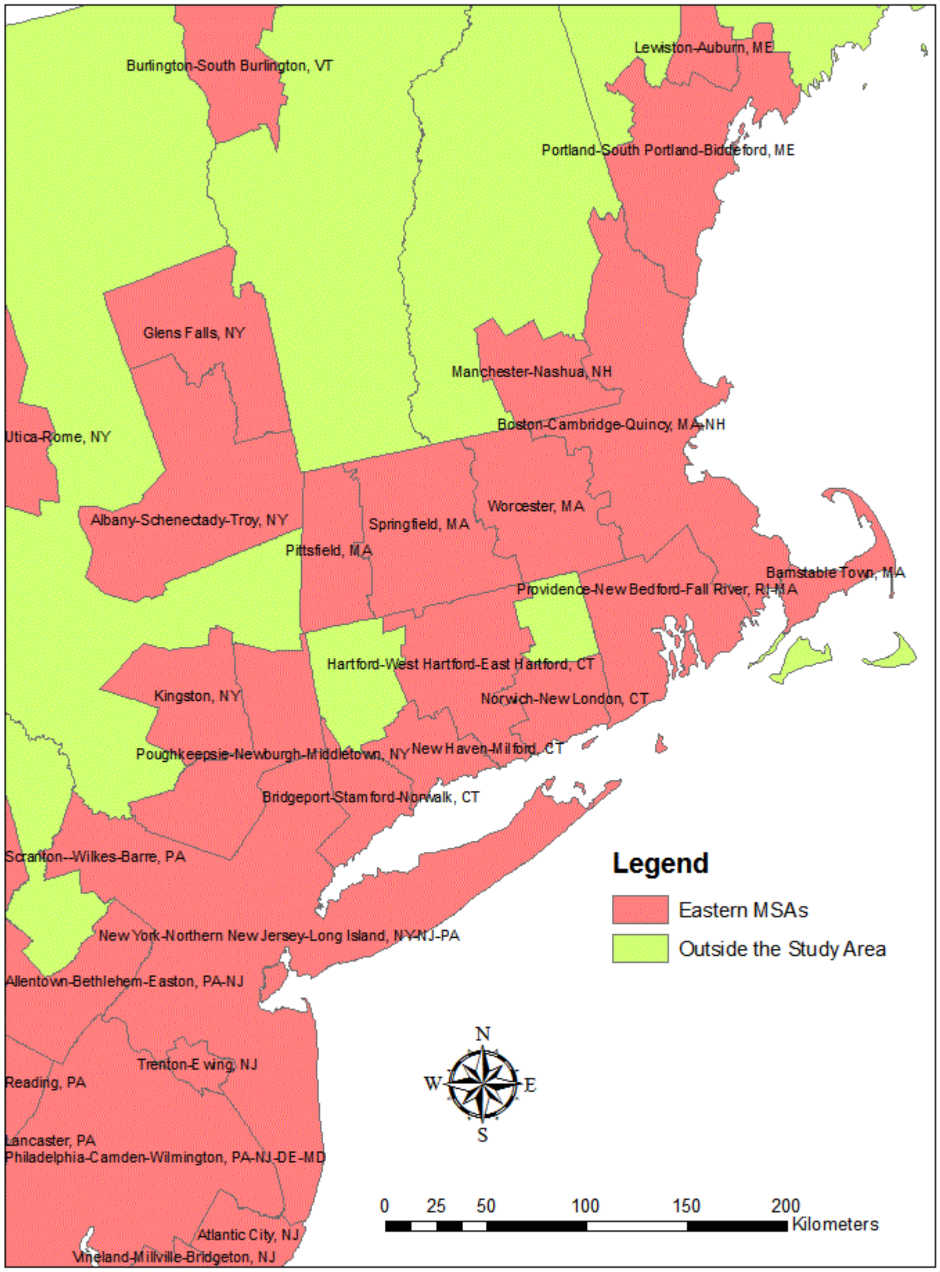

Figure 3.2: Urban regions (i.e., MSAs) in pink in the Boston-New York City area. Note, because MSAs are based on counties, they can include largely forested areas such as the Berkshire region of the western Massachusetts.

3.2.2 Data Sources

3.2.2.1 National Land Cover Database 2001

The National Land Cover Database 2001 (NLCD 2001) is a comprehensive land cover database for the all 50 states and Puerto Rico, produced cooperatively by the Multi-Resolution Land Characteristics Consortium (MRLC), which consists of 10 federal agencies, including the U.S. Geological Survey, Environmental Protection Agency, National Oceanic and Atmospheric Administration, the U.S. Forest Service, the National Atmospheric and Space Administration, the Bureau of Land Management, the National Park Service, the Natural Resources Conservation Service, the U.S. Fish and Wildlife Service, and the Office of Surface Mining Reclamation and Enforcement (Homer et al. 2004, 2007, U.S. Department of the Interior and U.S. Geological Survey 2008). Primary components of NLCD 2001 include: classified land cover data derived from imagery and ancillary data; ancillary data including a 30 m Digital Elevation Model (DEM), slope, aspect, and a positional index; and per-pixel estimates of percent imperviousness and percent tree canopy (Homer et al. 2004, U.S. Department of the Interior and U.S. Geological Survey 2008). The NLCD 2001 was created from nationwide Landsat 7 Enhanced Thematic Mapper Plus (ETM+) and Landsat 5 TM imagery (Homer et al. 2004). The minimum mapping unit of the land cover data is 0.4 ha (1 acre) (Homer et al. 2007). The resolution of the land cover data is 30 m and there are 16 land cover classes, excluding the land cover classes specific to coastal areas and Alaska (Homer et al. 2004, 2007). Training data were used to map all land cover classes except the “developed”, or urban, classes. The four “developed” classes were derived from thresholding of the imperviousness data product (Homer et al. 2007).

3.2.2.2 North American Breeding Bird Survey

The breeding bird data was acquired from the North American Breeding Bird Survey (BBS). The BBS is one of the most comprehensive and long lasting bird surveys since 1966, covering the entire U.S. and southern Canada. The BBS was started by Chandler Robbins and his colleagues at the Migratory Bird Population Station (now the Patuxent Wildlife Research Center) to monitor long-term trends among breeding birds (USGS Patuxent Wildlife Research Center 2008). Over 4100 survey routes are randomly located on secondary roads across the continental U.S. and Canada. Each survey route is 39.4-km (24.5-mile) long with stops at 0.8-km (0.5-mile) intervals (50 stops per route). At each stop, a 3-minute point count is conducted by trained volunteers and professionals once in a year typically during June or early July, depending on latitude (during the height of the avian breeding season). During the count, every bird seen within a 0.4-km (0.25-mile) radius or heard is recorded. Surveys start one-half hour before local sunrise and take about 5 hours to complete (USGS Patuxent Wildlife Research Center 2008). Survey data are accessible online at the BBS website (http://www.pwrc.usgs.gov/bbs/).

BBS's characteristic of being one of the most comprehensive, long-lasting, continental-scale bird surveys is suitable for my broad-scale, observational study. The BBS data can be analyzed to provide an index of population abundance (USGS Patuxent Wildlife Research Center 2008). The BBS data can also be used to estimate population trends along with other indicators to assess bird conservation priorities (USGS Patuxent Wildlife Research Center 2008). The BBS data is particularly useful when coupled with another broad-scale dataset, for example, the NLCD 2001, to analyze the relationship between long-term trends of bird populations and land cover change.

3.2.3. Bird Selection

The general trend of landscape change that is the concern of this research is the loss of forest patches in urban regions and the increase in forest edges due to the fragmentation of forests by suburban sprawl and to the conversion of forests to agricultural lands/residential uses. These changes result in the decline of forest interior species that require large interior area and the increase of edge species/generalist species because edge habitats are becoming more abundant (Freemark and Collins 1992, Twedt et al. 2006, Mason et al. 2007). Therefore, forest interior species is arguably a good indicator of the loss and fragmentation of quality forest habitat, which also provides other ecological functions (e.g., water retention, water purification, air purification, wildlife, timber). Because the declining forest interior species are of conservation concern (Whitcomb et al. 1981, Robbins et al. 1989), the following criteria were developed to select the bird species for detailed examination.

3.2.3.1 Selection Criteria

To select forest bird species that would respond favorably to the increase in the amount/connectivity of forest cover, bird species were selected based on the following criteria: (1) their numbers are declining, thus of conservation concern; (2) they are forest-interior/interior-edge species (or area-sensitive or fragmentation-sensitive species); (3) they are widely distributed across the eastern U.S. or a substantial portion of their range falls within the study area; and (4) they are neither too rare nor too common in urban areas. Ideally, the selected species would be good indicators of the loss and fragmentation of forest as habitat. The selected forest breeding bird species conform to the above criteria: Eastern Wood-Pewee (*Contopus virens*), Wood Thrush (*Hylocichla mustelina*), Black-and-white Warbler (*Mniotilta varia*), Ovenbird (*Seiurus aurocapillus*), and American Redstart (*Setophaga ruticilla*).

3.2.3.2 Selection Procedure

First, candidate bird species were selected based on the trend analysis, a program available within the BBS website (Sauer et al. 2005). The trend analysis program—based on population trend estimates, expressed as a percent change per year—allows users to quickly generate a list of species that showed either declining or increasing trend over specified periods based on breeding or nesting characteristics such as woodland breeding and open-cup nesting. This program was used to select the candidate species from the species in the "woodland breeding" category that showed *significant* ($p < 0.05$) declining trends over 1980-2005. The result of the trend analysis is shown in Table 3.1. The woodland breeding category was used because (1) the independent variables (in the following regression analysis) measure the landscape structure characteristics of forest cover or woodland and (2) the bird abundance data are taken during a breeding season. My assumption is that the species in this category would be most affected by the changes in forest cover structure such as fragmentation. The same trend analysis was also conducted for the bird species in the "open-cup nesting" category for reference since many species in this category are neotropical migrants who are more susceptible to forest fragmentation than those with other migratory habits (Whitcomb et al. 1981, Robbins et al. 1989). The result of this analysis is shown in Table 3.2.

As seen in Table 3.1, Eastern Wood-Pewee and Wood Thrush were estimated to be significantly declining in the majority of the 19 eastern states. (The trend estimates could not be produced for two states—Delaware and Rhode Island—because the birds were not observed in enough number of routes to accurately estimate the trend over time.) Black-and-white Warbler and Yellow-billed Cuckoo showed a significantly declining trend in more than 40% of the eastern states. American Redstart, Black-billed Cuckoo, Carolina Chickadee, Kentucky Warbler, Least Flycatcher, Ovenbird, Rose-breasted Grosbeak, and Veery showed a significant decline in more than 25% of the states. In sum, the woodland breeding species listed above showed a significant decline in many of the eastern states over the past 25 years.

Second, among these candidate forest breeding bird species, those that could potentially breed in the interior of a forest patch (both interior and interior/edge) were selected based on Freemark and Collins' (1992) classification. Also, abundance across the study area and habitat associations were used to further select the finalists. For example, some candidate species prefer particular woodland habitat such as swampy woodlands or forest edges; these species were judged not appropriate according to the criteria above. Coe's *Eastern Birds* (2001) was used as a reference to evaluate the candidate species for the appropriateness of the study based on their home range and preferred habitat. American Redstart was selected over Veery and Rose-breasted Grosbeak (two replacement candidate species) because (1) it has a more widespread summer breeding range and (2) although American Redstart's habitat includes secondary growth forest, Veery prefers disturbed and damp forest and Rose-breasted Grosbeak associates itself with edges, orchards, suburban parks and gardens (Cornell Lab of Ornithology 2008)—less appropriate based on the selection criteria. Using the criteria number 2, 3, and 4, Eastern Wood-Pewee, Wood Thrush, Black-and-white Warbler, Ovenbird, and American Redstart were selected for a final cross-examination.

Third, the above finalists were cross-examined by the earlier studies (e.g., Whitcomb et al. 1981, Robbins et al. 1989, Lee et al. 2002, and Vance et al. 2003) for their appropriateness to be used in my study. The result of the trend analysis of the species in the open-cup nesting group (Table 3.2) was also consulted. Because the earlier studies had similar study area and research interest (i.e., the relationship between the area and spatial configuration of forest cover and the abundance of forest bird species) as my study, by consulting these studies, the appropriateness of using the selected species for this study was hoped to be evaluated.

In sum, the selected forest (woodland) breeding species all showed a significantly declining trend in at least more than 25% of the 19 eastern states from 1980 to 2005. Taxonomically, they all belong to the same order of birds, Passeriformes (perching birds). All the selected species are neotropical migrants—according to Whitcomb et al. (1981), the single most important characteristics associated with forest fragmentation. In other words, the selected forest bird species belong to the same guild (species with similar life-history). Also, except for Ovenbird, which makes a dome-shaped nest, all the other selected species have open-cup type nests (Gough et al. 1998)—another indicator of susceptibility to forest fragmentation (Whitcomb et al. 1981, Lampila et al. 2005) because it is easier for Brown cowbirds, which increase in number as the forest edge increases due to the loss and fragmentation of forest, to lay eggs in open-cup nests, making open-cup nesters more susceptible to parasitism. With all these characteristics, the selected forest bird species can be expected to be a good indicator of forest loss and fragmentation.

3.2.4 Analysis Procedure

3.2.4.1 Overall Framework

ArcGIS v9.3 (ESRI 2009) was used for data storage, maintenance, and analysis. To investigate the research questions, two independent, continental-scale datasets (i.e., the NLCD 2001 and the BBS) were merged. By linking these data spatially, the information on bird relative abundances was coupled with land cover, enabling the analysis of landscape structure in the areas immediately surrounding each BBS survey route. Land cover data was acquired from the NLCD 2001; the original land cover classes were aggregated to seven land cover classes (one of them being forest cover class) to be applied to broad regions (see Table 3.3 for reclassification). The amount and spatial configuration (e.g., measures of connectivity/fragmentation) of forest cover were calculated within three buffers of varying width around each BBS route. The amount of forest was quantified as an average percent forest cover within each buffer from the land cover maps where the original forest cover classes (i.e., deciduous, coniferous, and mixed forest) were aggregated into a single "forest" cover class. FRAGSTATS, a computer software program to compute various landscape metrics for categorical map patterns (McGarigal and Marks 1995, McGarigal et al. 2002), was used to quantify the landscape structure around each BBS survey route.

The spatial configuration of forest cover around each survey route was quantified using FRAGSTATS. The percentage of each land cover class in each buffer width was also calculated to see what land cover affects the abundance of the selected forest bird species most. These variables (i.e., percent forest cover, percent land cover class, and various landscape structure metrics of "forest" cover) were used as predictor variables in a regression model to predict the bird abundance for each species.

FRAGSTATS is one of the most popular computer software programs used to quantify landscape structure. Since its development it has been widely used by many researchers to quantify landscape structure (composition and configuration) characteristics, and resulting landscape (structure) metrics have been used in landscape structure analysis in various geographic regions (e.g., Griffith et al. 2000, Li et al. 2001, Apan et al. 2002, Staus et al. 2002, Bender et al. 2003, Tinker et al. 2003, Kong and Nakagoshi 2006), land use/cover change studies (e.g., Li et al. 2004, Southworth et al. 2004, Weng 2007, Ma et al. 2008), and have been associated with the distribution and abundance of animals (e.g., Roseberry and Sudkamp 1998, Penhollow and Stauffer 2000, Grainger et al. 2005, Acevedo et al. 2006). Other applications include the development of landscape plans, comparison of the consequences of various simulation models and alternative future landscape scenarios, and association with residents' perception of scenic value (e.g., Gustafson 1998, Hulse et al. 2002, Leitão and Ahern 2002, Steinitz et al. 2003, Palmer 2004, Corry 2005). Specifically, FRAGSTATS has been used to analyze landscape structure around sampling points/plots in bird-habitat relationship studies (e.g., Robinson et al. 1995, Mörtberg and Wallentinus 2000, Penhollow and Stauffer 2000, Donovan and Flather 2002, Neel et al. 2004, Fearer et al. 2007, Caprio et al. 2009). Due to FRAGSTATS's wide application to landscape plans and ecological functions, along with the ease of consultation with the expert (one of the initial developers of the program) on my committee, I have decided to use FRAGSTATS to aid my analysis of landscape structure surrounding each BBS route.

3.2.4.2 Delineating Urban Regions

The Metropolitan Statistical Areas (MSAs), as urban regions, were acquired in ArcView Shapefile (.shp) format from the U.S. Census Bureau's Cartographic Boundary Files website (U.S. Census Bureau 2007a). State boundaries were drawn from the separate Census 2000 data that show states and state equivalent areas (e.g., the District of Columbia) in ArcView Shapefile format (U.S. Census Bureau 2007a). Both the MSAs and state boundary layers were projected, using the U.S.A. Contiguous Albers Equal Area Conic USGS version projection. The North American Datum of 1983 was used. Then, the 21 eastern states and the D.C. (i.e., the study area) were selected from the state map (Figure 3.1).

To select the MSAs that are in the study area, the MSAs that "intersect" the study area were selected. For those MSAs that have areas both outside and inside the study area, only the portion that lies within it was left to be included (Figure 3.3). The MSAs were used to select those routes that are in more urbanized areas. Note that since MSAs are based on counties, some MSAs include areas that are more rural than other MSAs that more densely populated. If necessary, additional criteria such as a certain percentage of imperviousness can be used in conjunction to select more urbanized MSAs as urban regions.

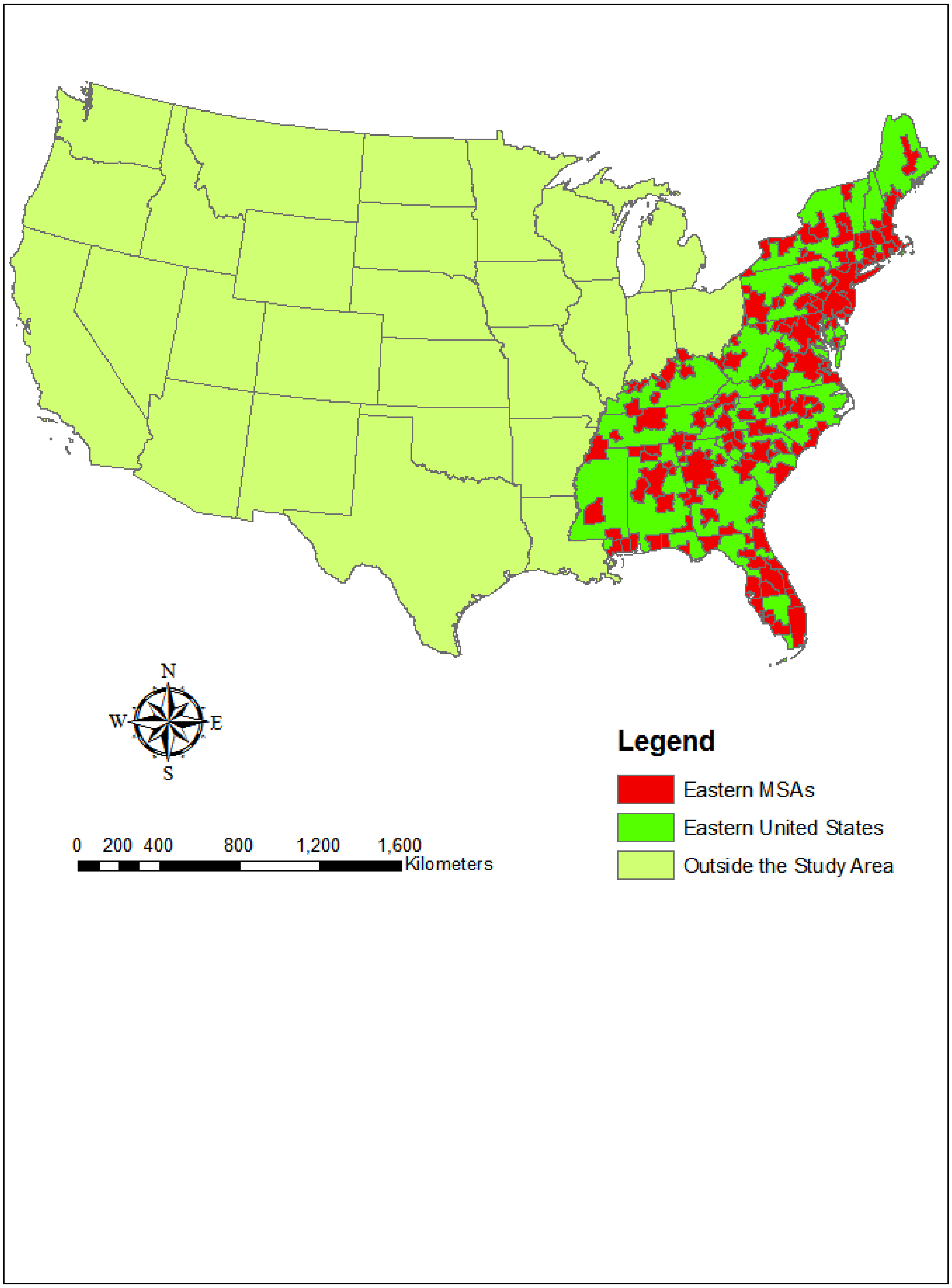

Figure 3.3: Urban regions (MSAs) are shown in red within the study area in darker green in the conterminous U.S.

3.2.4.3 Selecting the BBS Routes and Acquiring Bird Abundance Data

The BBS routes in the lower 48 States which were considered active in 1998 (USGS Patuxent Wildlife Research Center 2006) were downloaded from the National Atlas website (http://www.nationalatlas.gov/mld/bbsrtsl.html) and re-projected on top of the land cover maps. Each route has an identification number and a name by which it can be identified for the subsequent years to check whether or not it was active. Only the BBS routes that are mostly within the MSAs in the study area were selected, resulting in 402 routes. Using the BBS website (Sauer et al. 2005, 2006, 2007), for each of the selected routes, the abundance estimate (the average number of individuals) of the selected forest breeding bird species was recorded for 2002-2006. The years correspond to the years following the nominal year (2001) from which most of the satellite imagery used to develop the NLCD 2001 were acquired (Homer et al. 2007). If the routes were not active for at least two of the five years, they were removed from the dataset, resulting in 317 routes (Figure 3.4). To account for yearly fluctuations in bird abundance and the observer effect, for each route, the bird abundance was averaged for the available years. Average route-level abundance estimates over the minimum two years (and up to five years) are summarized in Appendix B.

As the bird data was carefully examined, it was found that none of the selected bird species is observed in many southern routes, especially the routes in Florida. Because the breeding ranges of the selected species do not extend as far south as Florida (Ridgley et al. 2005, Sauer et al. 2007), inclusion of these routes may lead to a spurious relationship between zero bird abundance and the landscape structure characteristics of the areas surrounding the routes that are actually outside the breeding ranges of the selected species. To minimize this possibility, only the BBS route data for all routes falling within the boundary of the species breeding distributions plus an additional 50 km buffer to account for uncertainty in estimating a species' true distribution were used for further analysis—following the procedure outlined in Fearer et al. (2007). For each of the five selected species, I used the geographic distribution maps provided by NatureServe in collaboration with Robert Ridgely, James Zook, The Nature Conservancy—Migratory Bird Program, Conservation International—Center for Applied Biodiversity Science, World Wildlife Fund—US, and Environment Canada—WILDSPACE (Ridgley et al. 2005). These maps were a compilation of range data from more than 46 different sources such as field guides and monitoring databases (Ridgley et al. 2005). From the geographic distribution maps, the breeding range for each species was selected. Then, a 50 km buffer was created around the breeding range. Finally, for each species, only the routes that are completely within the breeding range plus the 50-km buffer outside of the range were selected for further landscape structure analysis. In the end, 288 routes were selected for Eastern Wood-Pewee; 242, 245, 253, and 287 for Ovenbird, Black-and-white Warbler, American Redstart, and Wood Thrush, respectively. For the 6 km buffer distance (see below), the same size is a little smaller because those buffers that extent out of the study area were removed from the analysis. For the 6 km buffer distance, 281 routes were selected for Eastern Wood-Pewee (EAWP); 237, 240, 248, and 280 for Ovenbird (OVEN), Black-and-white Warbler (BWWA), American Redstart (AMRE), and Wood Thrush (WOTH), respectively.

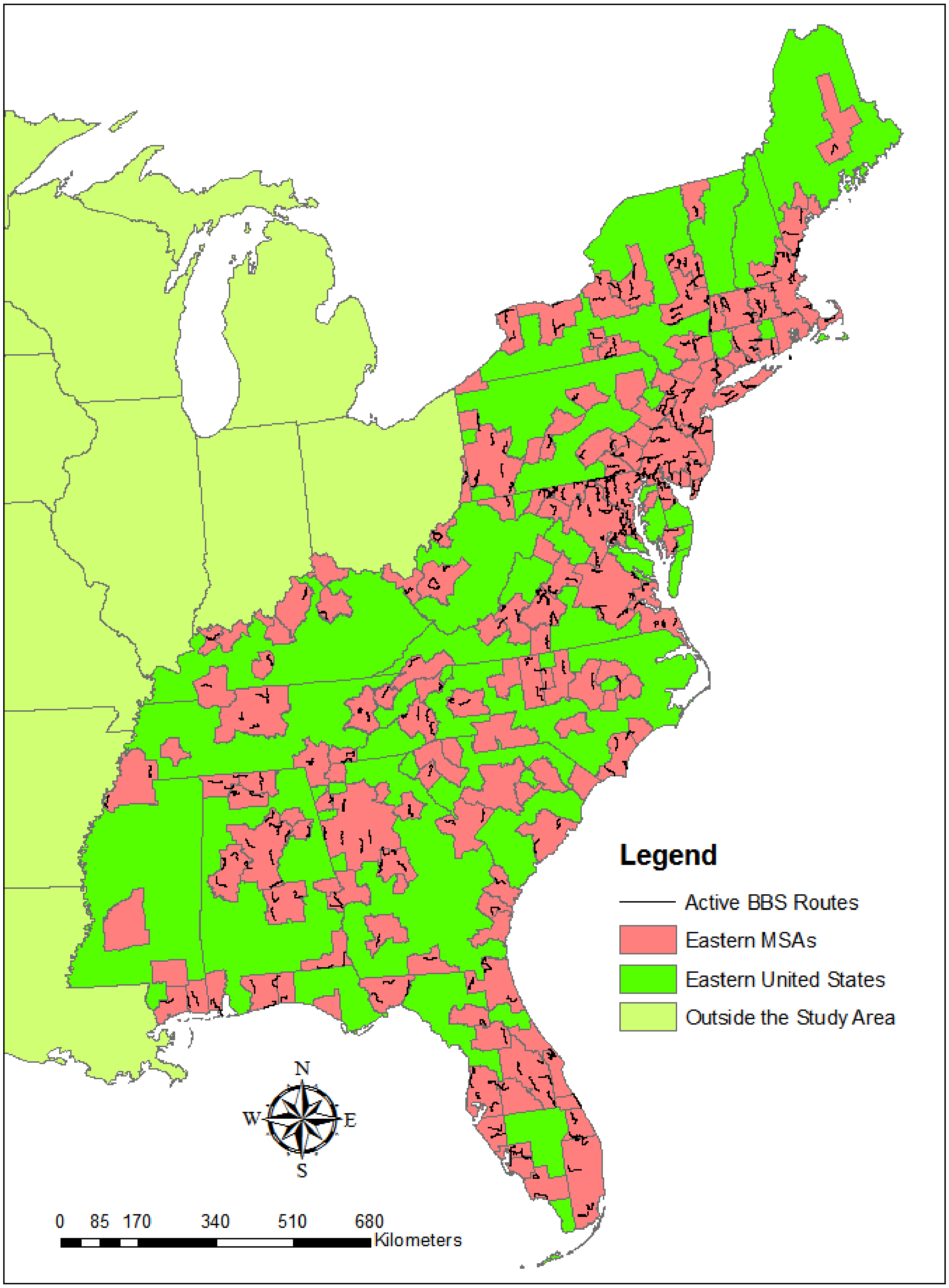

Figure 3.4: Selected active BBS routes in urban regions (MSAs) in the study area.

3.2.4.4 Buffers

It has been shown that habitat patch characteristics and the surrounding landscape context affect the abundance and distribution of organisms (Mörtberg 2001, Lee et al. 2002, Pidgeon et al. 2007, Radford and Bennett 2007, Martensen et al. 2008). Landscape structure characteristics in the areas around sample points/patches/transects (i.e., the surrounding landscape matrix) affect bird abundance. This influence can be investigated by establishing buffers at multiple scales around sample points/transects and quantifying the landscape structure within the buffers. For this study, buffers with three different widths of 180 m, 2010 m, and 6000 m (6 km) were created around each BBS route.

Three buffer distances were established because at least three nested scales are needed for an analysis of complex ecological phenomena according to hierarchy theory (Allen and Starr 1982, O'Neill et al. 1986, Freemark et al. 2002). These scales are: (1) the scale at which the phenomenon of interest occurs—in this study, the distribution and abundance of forest birds during breeding, (2) a finer scale than the target scale (in this study, the 180 m buffer distance), which explains the mechanisms of the target phenomenon, and (3) a coarser scale that governs the phenomenon at the target scale. Moreover, multi-scale analysis and exploration is necessary because we often do not know *a priori* at what spatial scale the birds are responding to landscape structure characteristics (Wiens 1989, Hostetler 2001, Thompson and McGarigal 2002), in particular, the amount and spatial configuration of forest. Creating varying buffer distances to demarcate a "landscape" around each survey route serves for multi-scale analysis (Thogmartin et al. 2004).

Most previous studies that used the BBS as the source of bird abundance and occurrence established a circle of radius 19.7 km (12.24 miles) centered on each BBS route for landscape structure analysis (e.g., Flather and Sauer 1996, Boulinier et al. 2001, Donovan and Flather 2002, Vance et al. 2003, Pidgeon et al. 2007). This radius would completely contain the route even if it were a straight line. I argue that the radius of the circle has little ecological meaning except that it may coincidently relate to the mean maximum natal dispersal distance (20.5 km) of 10 forest-nesting, neotropical migrant species (Donovan and Flather 2002). As for the other bird-habitat relationship studies that did not use the BBS as the source of bird data, earlier studies tended to focus on the characteristics (e.g., patch size, the perimeter-area ratio of the patch) of the habitat patch itself in which a sample point is located and tended not to establish an area around a sample point. When some studies did establish a "landscape" around a sample point/study plot, 2 km (1.24 miles) was a commonly-used, maximum radius (e.g., Robbins et al. 1989, Betts et al. 2006b, Mattsson and Niemi 2006, Betts et al. 2007a). Percent forest cover within a landscape, a circle of some large radius (e.g., 2, 10, 30 km) from the sample point, was commonly calculated to quantify forest isolation (fragmentation) (Robbins et al. 1989, Robinson et al. 1995, Rosenberg et al. 1999, Veech 2006).

In the end, three varying buffer distances were decided: 180 m, 2010 m, and 6 km. Since the pixel size is 30 m, these values are closest multiples of 30 of the intended values: 193 m, 2000 m, and 6 km, respectively. Since a buffer is established around the entire length of a BBS route, not including the route in a buffer is not an issue. 193 m was chosen because it is the maximum breeding territory size (radius) of the five species

(Askins and Philbrick 1987, Yahner 1993). 180 m (591 ft) was included as the closest multiple of 30 m, and because the average breeding territory size is smaller than 193 m (Whitcomb et al. 1981, Askins and Philbrick 1987, Yahner 1993). 2010 m corresponds to the mean flight distance (2100 m) in a series of flights that Wood Thrush made during homing experiments when they were breeding (Able et al. 1984). 2000 m also coincides with the average juvenile dispersal distance of Wood Thrush (Anders et al. 1998, Rivera et al. 1998, Lang et al. 2002). In addition, as noted above, 2000 m (1.24 miles) is an often used maximum radius to establish a “landscape” around each sample point. 6 km was decided as the largest “landscape” around the survey routes based on hierarchy theory that at least three scales are necessary to understand and analyze ecological processes. The 6 km buffer contains the target scale, 2000 m (2 km), giving constraints to the phenomena at the target scale. The largest landscape is used mainly to study the effects of the surrounding landscape (i.e., matrix) on forest patches included within (e.g., Stouffer et al. 2006). The largest landscape is also needed to counter the limitation that for Isolation/Proximity metrics (to calculate nearest-neighbor distances) and Connectance metric, only patches within the landscape are used to calculate them even when in fact a patches’ nearest neighbor may be just outside the landscape boundary (McGarigal et al. 2002). In short, the three buffer distances are ecologically meaningful for the selected forest breeding bird species.

3.2.4.5 Reclassification of Land Cover Maps

Land cover maps are one of the primary components of NLCD 2001. Four zonal maps (zones 11, 12, 13, and 14) covering the study area were download from the MRLC website (http://www.mrlc.gov/nlcd_multizone_map.php). The original 16 land cover classes were aggregated to seven functional land cover classes: unvegetated, open space, low imperviousness, high imperviousness, forest, shrub, and herbaceous (Table 3.3). The classes were developed based on vegetation cover characteristics and their expected contribution to act as the birds' habitat. Note that forest land cover class consists of original deciduous, evergreen, and mixed forest cover classes.

3.2.4.6 Landscape Structure Measures

The following 14 landscape structure metrics were computed from the reclassified land cover map using FRAGSTATS version 3.3 (McGarigal et al. 2002): (1) for all land cover types, percentage of landscape (PLAND), Simpson's diversity index (SIDI); (2) for aggregated "forest" land cover type, patch density (PD), area-weighted mean radius of gyration (GYRATE_AM), area-weighted mean shape index (SHAPE_AM), area-weighted mean fractal dimension index (FRAC_AM), perimeter-area fractal dimension (PAFRAC), area-weighted mean proximity index (PROX_AM), area-weighted mean similarity index (SIMI_AM), area-weighted mean Euclidean nearest-neighbor distance (ENN_AM), contrast-weighted edge density (CWED), patch cohesion index (COHESION), connectance index (CONNECT), and contagion (CONTAG). Area-weighted mean was chosen as a way to integrate over all the patches of forest cover type to place an emphasis on the effect of large patches that are known to affect forest bird abundance. The selection of the landscape structure metrics was based on their reported association with the distribution and abundance of birds (van Dorp and Opdam 1987, Donovan et al.1995, Robinson et al. 1995, Flather and Sauer 1996, Mörtberg and Wallentinus 2000, Boulinier et al. 2001, Donovan and Flather 2002, Vance et al. 2003, Fearer et al. 2007, Pidgeon et al. 2007). The landscapes metrics, computation, and interpretation are summarized in Table 3.4.

In FRAGSTATS terms, SIDI and CONTAG are “landscape” metrics and the rest are “class” metrics. Landscape metrics are computed over the entire land mosaic, for all land cover types in a defined area (landscape). On the other hand, class metrics are computed for a specific land cover type but across all the patches belonging to that land cover type in a specified area (landscape). In another classification scheme, PLAND, PD, and SIDI are landscape composition metrics. Landscape composition refers to features associated with the variety and abundance of patch types within the landscape without reference to spatial attributes (McGarigal et al. 2002). The rest of the metrics are landscape configuration metrics, representing the spatial character and arrangement, position, or orientation of patches within the class or landscape (McGarigal et al. 2002).

The caveat in interpreting these metrics is that due to the different shape of a BBS survey route, even with the same buffer distance, the total buffer area varies in size. This is different from a constant one-big-circle approach of most of the previous studies to delineate a landscape (see, for example, Flather and Sauer 1996, Boulinier et al. 2001, Donovan and Flather 2002, Vance et al. 2003, Pidgeon et al. 2007). To avoid the area effect on bird abundance, for all the area-related measures, the percentage of a specific land cover type was used instead of the total area. The same reason applied to the decision of selecting density measures instead of a total number.

PLAND is percent of area occupied by each land cover type in a specified buffer size. Because a total buffer area is variable even for a same buffer distance, PLAND is a more objective measure of the contribution of each land cover type than the total area of each land cover type (i.e., total class area). PLAND increases as more areas in the total buffer area are occupied by the corresponding land cover type.

PD equals the number of forest patches divided by the total area of a buffer. Because the total buffer area is variable even for a same buffer distance, PD is a more objective measure than the number of forest patches, which increases by chance alone as the total area becomes larger. Therefore, PD facilitates comparisons among landscapes of varying size. Note that I use the 8-neighbor rule to define a patch.

Among the selected landscape metrics, those whose sole purpose is to measure connectivity include: COHESION, CONNECT, and GYRTAE_AM. COHESION, for this study, measures physical connectedness of the forest cover type. COHESION approaches 0 as the proportion of the landscape comprised of forest cover decreases and forest patches become increasingly subdivided and less physically connected. COHESION increases as the percentage of forest cover increases and forest patches become more clumped or aggregated in their distribution; hence, more physically connected.

CONNECT is also a measure of connectivity. CONNECT is defined as the number of functional joinings between patches of the corresponding patch type, where each pair of patches is either connected or not based on a user-specified distance criterion (i.e., a threshold distance). The threshold distance could be scaled to a functional distance. CONNECT is reported as a percentage of the maximum possible connectance given the number of patches. For this study, larger CONNECT values mean that more forest patches are within the threshold distance (2,000 m). A limitation of CONNECT is that only patches within the landscape are considered when determining if a patch is connected or not, even if a patches' nearest neighbor may be just outside the landscape boundary. Therefore, this is a serious limitation for this metric when the buffer size is 180 m.

GYRATE_AM is the same as correlation length, which is based on the average extensiveness of connected cells (i.e., patches). It is a measure of patch compaction. For this study, I am particularly interested in the connectedness of forest patches. GYRATE_AM is computed as the area-weighted mean radius of gyration across all forest patches in a certain buffer size. For this study, larger GYRATE_AM values mean that on average, forest patches are more extensive.

There are other FRAGSTATS metrics that indirectly measure connectivity. For example, ENN, PROX, and SIMI measure isolation of the patches of the focal class. Isolation deals explicitly with the spatial and temporal context of patches, rather than the spatial character of the patches themselves (McGarigal et al. 2002). ENN and PROX adopt an island biogeographic perspective on patch isolation; SIMI adopts a landscape mosaic perspective on patch isolation (McGarigal et al. 2002). The three metrics represent physical connectivity but SIMI is scaled for functional connectivity by a similarity coefficient between different land cover types.

ENN is the simplest, most direct measure of patch isolation. ENN is the Euclidean nearest-neighbor distance from the focal patch. ENN is computed for each forest patch and ENN_AM, as a distribution measure of ENN, equals the sum, across all forest patches, of ENN value multiplied by the proportional abundance of each patch.

PROX is a unitless measure of patch isolation that accounts for the size and distance of like patches from a specified focal patch within a defined search radius

(Leitão et al. 2006). A search radius or “neighborhood” for each focal patch corresponds with the particular organism or ecological process of interest. For example, a search radius can be set to match a home rage of the target organism. PROX is calculated for each patch by summing the area of each neighboring patch of the same land cover type as the focal patch that lies within a specified search radius from the focal patch, after weighing the area of each neighboring patch by its distance from the focal patch (Leitão et al. 2006). Larger values of PROX mean that neighboring patches of the same land cover type are larger and closer together (i.e., patches are less isolated). Smaller values indicate that patches are further apart and may be smaller in area (i.e., patches are more isolated). PROX is calculated at the patch level and can be summarized at the same class and landscape levels. In this study, area-weighted mean proximity (PROX_AM) at the class level is used to incorporate information about the relative importance of each patch based on its size into the degree of isolation (and fragmentation) of forest cover.

PROX is useful for comparing different patches within a landscape (e.g., which patches should have a higher priority of protection as nature reserves) or comparing the spatial configuration of patches in different landscapes (e.g., as used in my study) (Leitão et al. 2006). The major limitation of PROX as a measure of patch isolation is the use of Euclidean distances. The intervening matrix or land covers/uses, which may actually play a large role in effective isolation (by impeding the movement/dispersal of the organism of interest, for example), do not affect the value of PROX (Leitão et al. 2006).

SIMI operates just like PROX but each patch is weighted by its similarity to the focal patch. To compute SIMI, a similarity coefficient (0-1) is needed. For this study, a similarity coefficient is calculated for each pair of land cover types based on the average

% canopy for each cover type. (% canopy data are one of the auxiliary data of the NLCD 2001.) For example, forest cover type has on average 80% canopy cover; shrub cover type has on average 64% canopy cover (Table 3.5). In this case, the similarity coefficient between these two land cover types is calculated as: 64/80 = 0.8 (very similar). In another example, the similarity coefficient between open space (average 25% canopy) and low impervious cover (average 9% canopy) is: 9/25 = 0.36 (not very similar). Similarity coefficients were entered into a similarity weight file, which was used to compute SIMI. SIMI increases when the land cover types of the patches within the search radius become more similar (in terms of % canopy for this study) and those similar patches become closer and more contiguous in distribution. When computing both SIMI and PROX, only patches contained within a specified buffer size are considered in the computations when the search radius extends beyond the buffer (landscape) boundary. The difference between SIMI and PROX is that while SIMI considers all land cover types, PROX only considers forest cover type for this study.

SHAPE and FRAC are both measures of overall shape complexity. SHAPE equals patch perimeter (given in number of cell surfaces) divided by the minimum perimeter (given in number of cell surfaces) possible for a maximally compact patch of the corresponding patch area. FRAC equals 2 times the logarithm of patch perimeter divided by the logarithm of patch area. PAFRAC also reflects shape complexity across a range of patch sizes.

CWED equals the sum of the lengths of each forest edge segment multiplied by the corresponding contrast weight, divided by the total buffer area, multiplied by 10,000 (to convert to hectares). For this study, edge contrast between forest cover type and all the other land cover types is of interest. Edge contrast was considered to be the opposite of similarity in terms of average canopy % for different land cover types (Table 3.5). Contrast weights were developed by taking 1 minus the similarity coefficient for each pair of land cover types. For example, the contrast weight between forest cover and shrub cover is: 1-0.8 = 0.2 (low contrast). CWED increases as the amount of forest edge in the buffer increases and/or as the contrast in edges involving forest cover type increases (i.e., the contrast weight approaches 1). By computing density, it facilitates comparison among landscapes (buffers) of variable sizes.

SIDI and CONTAG are landscape-level metrics. SIDI equals 1 minus the sum, across all patch types, of the proportional abundance of each patch type squared. It represents the diversity (number and evenness) of the reclassified land cover types in the buffer. SIDI represents the probability that any 2 cells selected at random would be different land cover (patch) types (McGarigal et al. 2002). SIDI approaches 1 as the number of different land cover types increases and the proportional distribution of area among land cover types becomes more equitable.

CONTAG calculates the probability that two randomly chosen adjacent cells belong to the same land over class. CONTAG equals 1 minus the sum of the proportional abundance of each patch type multiplied by the proportion of adjacencies between cells of that patch type and another patch type, multiplied by the logarithm of the same quantity, summed over each unique adjacency type and each patch type; divided by 2 times the logarithm of the number of patch types; multiplied by 100 (to convert to a percentage). CONTAG increases when patches become more aggregated (i.e., more like-cell adjacencies) and less interspersed (i.e., inequitable distribution of pairwise adjacencies). CONTAG is inversely related to edge density (McGarigal et al. 2002).

2000 m was used as the threshold distance to calculate CONNECT and as the search radius to calculate isolation/proximity metrics such as PROX, SIMI, and ENN. This is the distance which the selected birds perceive to be connected (functional connection), or perceive that they can easily move between (similar to an ecological neighborhood) (McGarigal et al. 2002). 2000 m was chosen as the threshold distance and the search radius because: (1) birds are more vagile than ground-crawling, small mammals, not to mention amphibians and insects; (2) all the selected forest birds are neotropical migrants, so they are capable of long-distance flights; and (3) Able et al. (1984) reported that breeding Wood Thrushes moved in a series of short flights (mean = 2100 m) during homing experiments, and 2000 m also coincides with the average juvenile dispersal distance of Wood Thrush (Anders et al. 1998, Rivera et al. 1998, Lang et al. 2002). A caveat in computing these metrics is that if the threshold or search radius extends outside the buffer size, the indices are computed within the buffer. Therefore, for the 180 m buffer size, these indices are computed within the buffer size.

In sum, at each buffer distance, three categories of measures were taken from each buffer: percent land cover class, landscape structure measures of forest land cover, and landscape-level metrics based on the entire land mosaic. The proportion of each land cover class (after reclassification) in each buffer was calculated to see which land cover affects the bird abundance most. The focus is on landscape structure metrics of forest cover with particular emphasis on the spatial configuration of forest cover (e.g., measures of connectivity/fragmentation). For any area-based measures, because each buffer area was variable even with the same buffer distance, due to the variable shape of a BBS route, percentages were used for comparison.

3.2.4.7 Statistical Analysis

3.2.4.7.1 Regression

The response variable was the mean (of the minimum two and the maximum five years) route-level abundance (i.e., the average of all the observations for each route) of the selected forest bird species. Average route-level abundance estimates were square root transformed to correct the distribution of the residuals of the regression (bird abundance against % forest or all the variables) so that it may become more normal. Overall, the transformation improved the distribution to be more normal. First, simple linear regression was conducted on the transformed bird abundance against % forest in the three buffer distances for each species.

Second, for each species at each buffer size, multiple regression analysis was conducted to investigate the correlation between bird abundance and various landscape structure metrics derived by FRAGSTATS. A full, additive model had 14 explanatory variables: in FRAGSTATS terms, 12 forest class-level metrics and two landscape-level metrics (see Table 3.4 for the list of variables and their interpretation). Therefore, the additive multiple linear regression model is:

(square-root transformed bird abundance) = β_0 + β_1(PLAND) + β_2(PD) + β_3(GYRATE_AM) + β_4(SHAPE_AM) + β_5(FRAC_AM) + β_6(PAFRAC) + β_7(PROX_AM) + β_8(SIMI_AM) + β_9(ENN_AM) + β_{10}(CWED) + β_{11}(COHESION) + β_{12}(CONNECT) + β_{13}(SIDI) + β_{14}(CONTAG) + ε_i

The goal of variable selection is to find a parsimonious subset of variables that has as low Akaike's Information Criterion (AIC) and as high adjusted R^2 as possible. Because the reduced model that gives the lowest AIC and/or the highest adjusted R^2 value may not necessarily have the fewest number of variables, balancing these objectives can at a time become more art than science. First, to check for multicollinearity among the variables Pearson product-moment bivariate correlations between pairwise combinations of variables were computed. Any pair with the magnitude of the correlation coefficient greater than 0.7 was marked as having a high correlation. Second, for each pair of variables identified as having a high correlation, redundancy analysis and partial redundancy analysis were conduced within 12 class-level metrics and between two landscape-level metrics separately. Comparing all the pairs of variables with a high correlation, the variables with more redundancy (the variable with the smaller marginal effect) and lower partial contribution (the variable with the smaller conditional effect) were removed. When the results of redundancy analysis and partial redundancy analysis did not agree, both variables in the pair were kept. Third, stepwise variable selection procedure was conducted with the remaining variables, noting AIC values. Both forward and backward selection procedures were tried to see if the resulting models which give the smallest AIC value agree. The model with the smaller AIC value was chosen. Fourth, the variance inflation factor (VIF) was calculated for the variables in the selected model to make sure that there was no collinearity problem. VIF values greater than 10 suggest strong collinearity. Finally, relatively more important variables among the variables in the final reduced model were determined using relaimpo (Grömping 2006) package. Relaimpo (Grömping 2006) package computes the relative contribution of each variable in the final reduced model to explaining the variation in bird abundance with CIs created by 1000 bootstrap replicates.

3.2.4.7.2 What Land Cover Affects Bird Abundance Most?

The percentage of each land cover type in each buffer size was calculated. Then, for each species at each buffer distance, the correlation between bird abundance and the percentage of each land cover type was calculated to investigate which land cover type had the most effect on the abundance of the selected forest bird species. Also, at each buffer size, pairwise correlation coefficients were calculated between forest cover type and the other reclassified land cover types to see if any land cover type was significantly correlated with forest cover type.

3.2.4.7.3 Threshold Detection

To see whether or not a threshold forest amount exists, first, the scatterplots of square-root transformed bird abundance against % forest were visually inspected for thresholds. Second, a local smoothing function (i.e., lowess line) was fit to the data. The lowess line was used to estimate where thresholds may lie. Third, piecewise regression models were fit to the data, using the piecewise linear model in the package segmented version 0.2-4 (Muggeo 2008).

Piecewise linear regression is a form of regression that allows multiple linear models to be fit to the data for different ranges of an explanatory variable, *x* (Toms and Lesperance 2003, Ryan and Porth 2007). Breakpoints are the values of *x* where the slope of the linear function changes. A breakpoint was defined here as the percentage of forest cover in a buffer where the fitted functions intersected. This was interpreted as the threshold of forest amount (loss) at which the relationship between bird abundance and % forest changes drastically.

3.3 Results

All the species had a right-skewed distribution with many average route-level abundance estimates toward 0s and tapering off to large values and some extremely large values (Figure 3.5). The means were low for all the species. The distribution of the percentage of forest cover (% forest) at the 2010 m buffer size (n = 291) showed a long gradient and was normal (Figure 3.6).

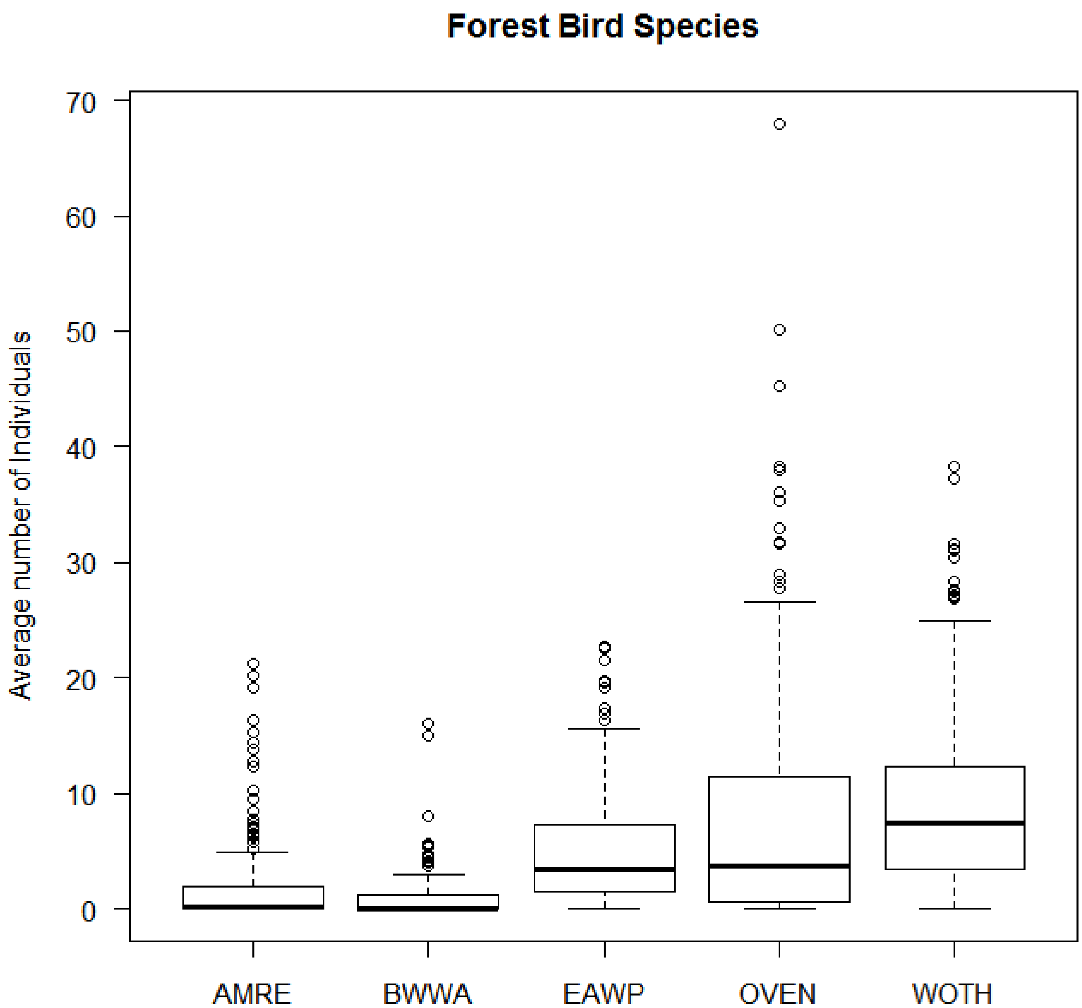

Figure 3.5: Right-skewed distributions shown in the box and whisker plots. The Y axis is the average route-level abundance over minimum two years between 2002 and 2006. There are some large numbers for each species, which are checked not being data entry errors.

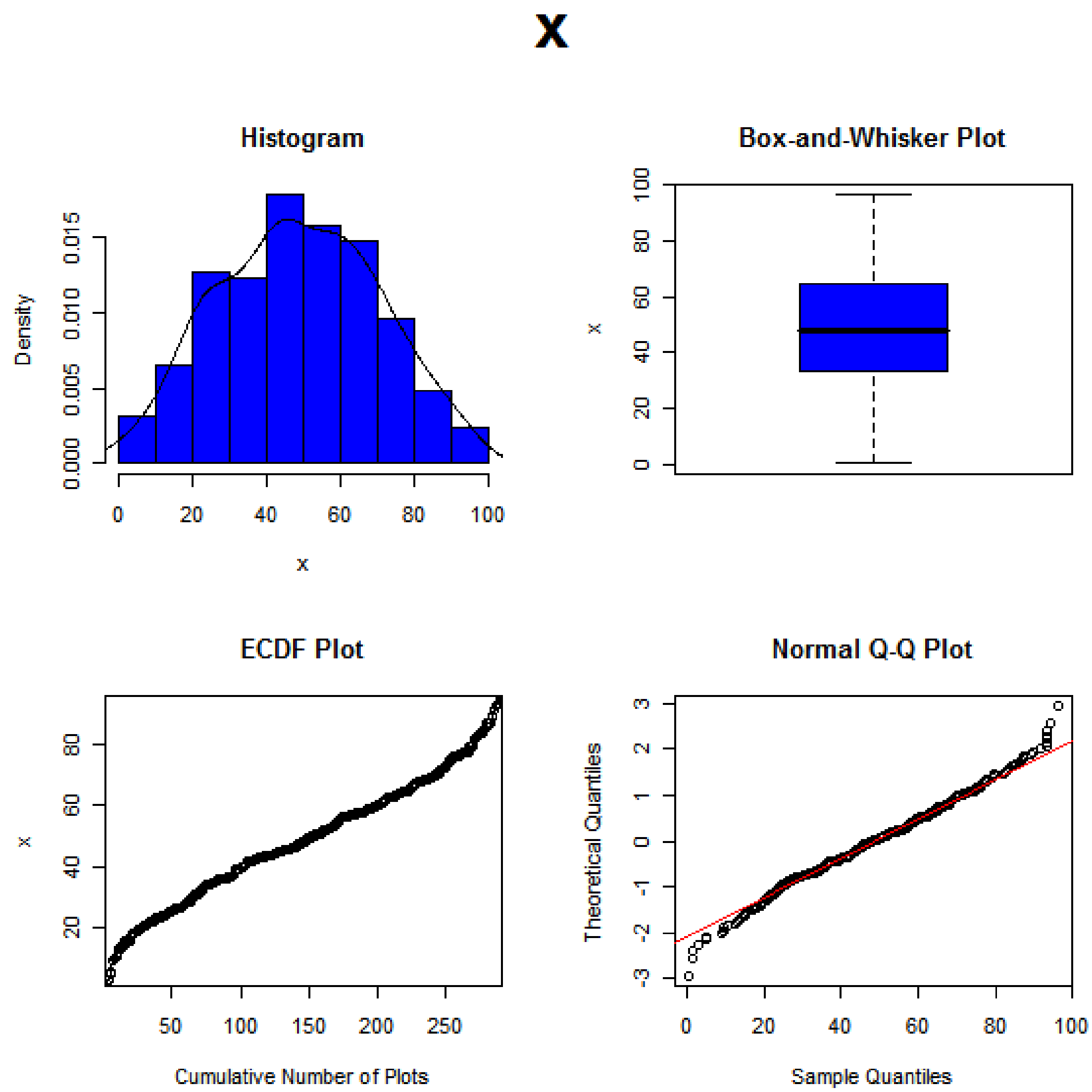

Figure 3.6: A long % forest gradient at the 2010 m buffer size. The distribution is roughly normal.

3.3.1 Simple Linear Regression

The results of simple linear regression analysis were almost identical across scales (Tables 3.6, 3.10, 3.14, 3.19). Either OVEN's or BWWA's abundance had the highest correlation with % forest with R^2 of around 0.30. AMRE always had the third highest correlation with % forest with R^2 of around 0.20, followed by WOTH's 0.08. A representative result at the 2010 m buffer size is shown in Figure 3.7. EAWP's abundance was always least correlated with % forest with R^2 of around 0.05. In all models, % forest was a statistically significant predictor of bird abundance (p-value < 0.05). The model slope estimates were all positive and none of the 95% confidence intervals (CIs) included 0. The intercepts of the simple linear regression models for EAWP and WOTH were always positive, whereas those for OVEN, BWWA, and AMRE were negative across scales.

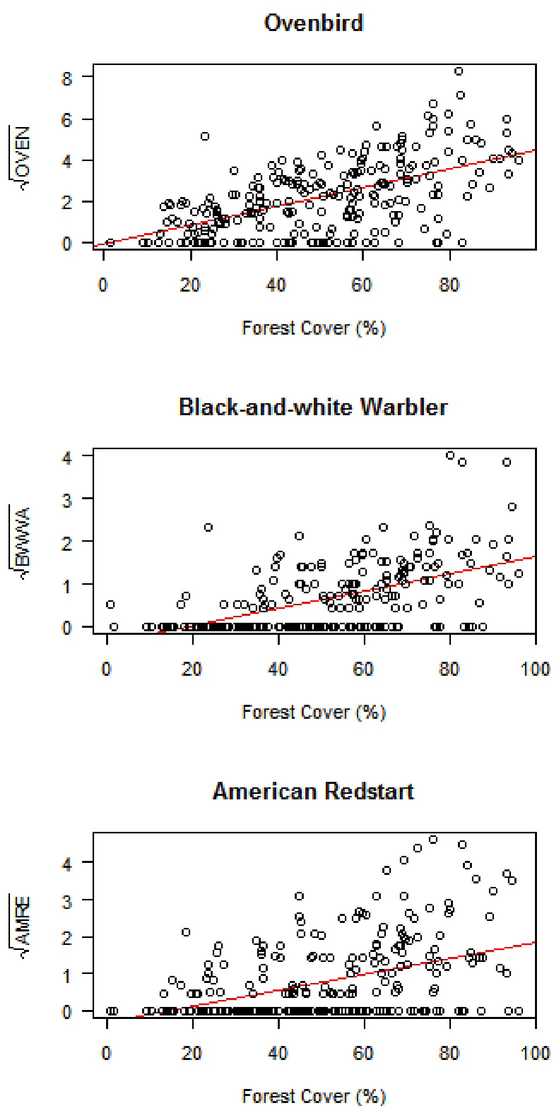

Figure 3.7: The correlation between the square-root transformed bird abundance and the percentage of forest cover in the 2010 m buffer. The red line is the simple linear regression line. The five species are placed in the order of decreasing R^2 value from left top, to left bottom, and right top to bottom. The first column has species with relatively high R^2 values; the second column has species with relatively low R^2 values.

3.3.2 What Land Cover Type Is the Best Predictor of Forest Bird Abundance?

The pattern of correlation between different land cover types and bird abundance was similar across scales (Tables 3.9, 3.13, and 3.17). For OVEN, BWWA, and AMRE, forest cover type explained the most variation in bird abundance, followed by herbaceous cover type. On the other hand, for EAWP and WOTH, bird abundance was not well correlated with forest cover type. For EAWP, other land cover types such as open space, low imperviousness, and high imperviousness explained much more variation in bird abundance than forest cover. For WOTH, either shrub cover or high imperviousness explained the most variation in bird abundance and forest cover was the second. Note that herbaceous cover was most correlated, albeit modestly, with forest cover.

3.3.3 Multiple Regression

In the additive, full multiple regression models, the 14 variables explained the most variance (> 40%) in OVEN, then BWWA (> 37%) across scales (Table 3.20). The variables always explained the least variance (< 26%) in WOTH (Table 3.20). The trend across scales was similar for the reduced multiple regression models. The selected subset of variables always explained the most variance in either OVEN or BWWA (Table 3.21). WOTH always had the lowest adjusted R^2 value (Table 3.21).

As for the important variables in the reduced models across scales, for OVEN and BWWA, PLAND (+) always contributed most to explaining the total variation in bird abundance (Table 3.22). SIMI_AM (+) and CWED (-) were the second most important variables. (The sign in the parenthesis indicates the sign of the variable's partial regression coefficient.)

SIDI (-) and CWED (+) were important predictors for WOTH (Table 3.22), which consistently had the lowest R^2 and adjusted R^2 values for the full and the reduced models across scales. Similarly, PLAND (+), CONTAG (+), CWED (+), and CWED (-) were important predictors for EAWP and AMRE, which had either the second or third lowest R^2 values for the regression models across scales. CWED (+), SIDI (-), CONTAG (+), CWED (-), and PLAND (+) were identified as important predictors for WOTH, AMRE, and EAWP across scales (Table 3.22).

At each scale in the reduced models, PLAND (+) was always selected as having the highest relative contribution to explaining the variance in bird abundance for more than one species (Table 3.23). CWED (+) and CWED (-) were also important predictors of bird abundance across scales. At the 2010 m buffer size, SIMI_AM (+) was also selected as important in addition to PLAND (+), CWED (+), and CWED (-).

3.3.4 Threshold Detection

As for the existence of percent forest cover thresholds, there were no clear thresholds that were consistently identified by the three methods of threshold analyses: a quick visual inspection of the scatterplots, lowess lines, and the piecewise linear regression model available in the package segmented. For all the species, a visual inspection of the scatterplots revealed no clear thresholds at any scale. Some thresholds, although mostly not clear, were suggested by lowess lines (Figure 3.8). These values were used as “seeds” to search thresholds in the subsequent piecewise linear regression analysis.

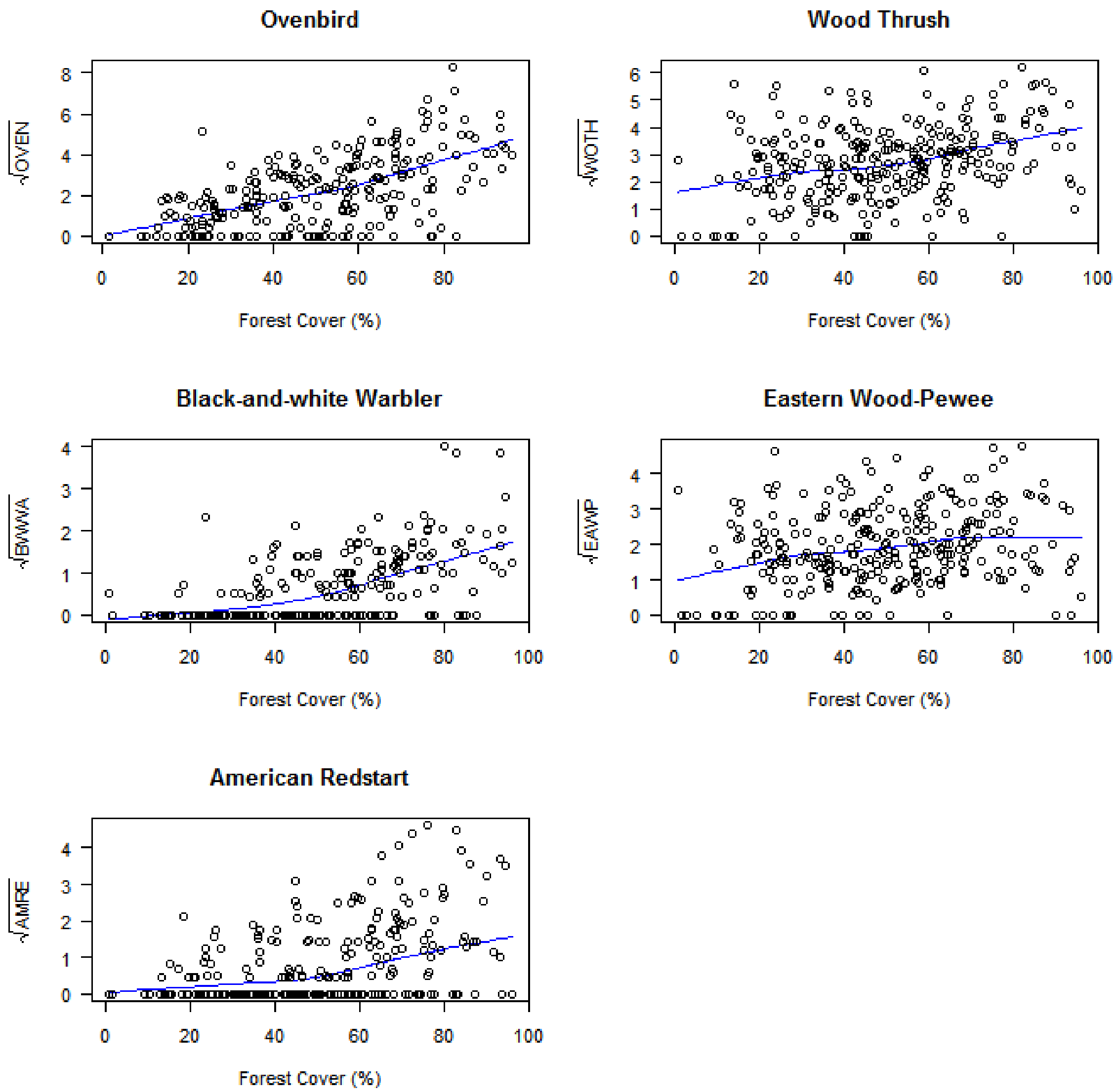

Figure 3.8: The lowess line (the blue line) fit through the scatterplot of square-root transformed bird abundance against the percentage of forest in the 2010 m buffer. The species are ordered in the decreasing R^2 value of the simple linear regression line from the top left column to the bottom left, to the top right, and to the bottom right column.

The piecewise linear regression models available in the package segmented identified thresholds for the one-breakpoint model for all species at all scales except for OVEN at the 6 km buffer size. However, most of the CIs of the identified thresholds were wide and some even contained 0. The only thresholds with narrow CIs were: for BWWA at the 180 m buffer size (87% forest cover) (Figure 3.9) and at the 6 km buffer size (86% forest cover) (Figure 3.10); for WOTH at the 180 m buffer size (9% forest cover) (Figure 3.11, Tables 3.8 and 3.16). For the two-breakpoint model, few stable thresholds were identified. Those thresholds with narrow and non-overlapping CIs were: for WOTH, 74% and 90% forest cover at the 2010 m buffer size (Figure 3.12), and 75% and 88% forest cover at the 6 km buffer size (Figure 3.13); for BWWA, 24% and 86% forest cover at the 180 m buffer size (Figure 3.14); and for AMRE, 36% and 71% forest cover at the 180 m buffer size (Figure 3.15, Tables 3.8, 3.12, and 3.16). Because the adjusted R^2 values of the two-breakpoint models that produced these thresholds were higher than the R^2 values of the corresponding simple linear regression models, the two-breakpoint models were considered to be better models. The rest of the identified thresholds had problems with (1) two thresholds being too close to each other, (2) the CIs being wide and overlapping, and/or (3) one of the CIs containing the other threshold value.

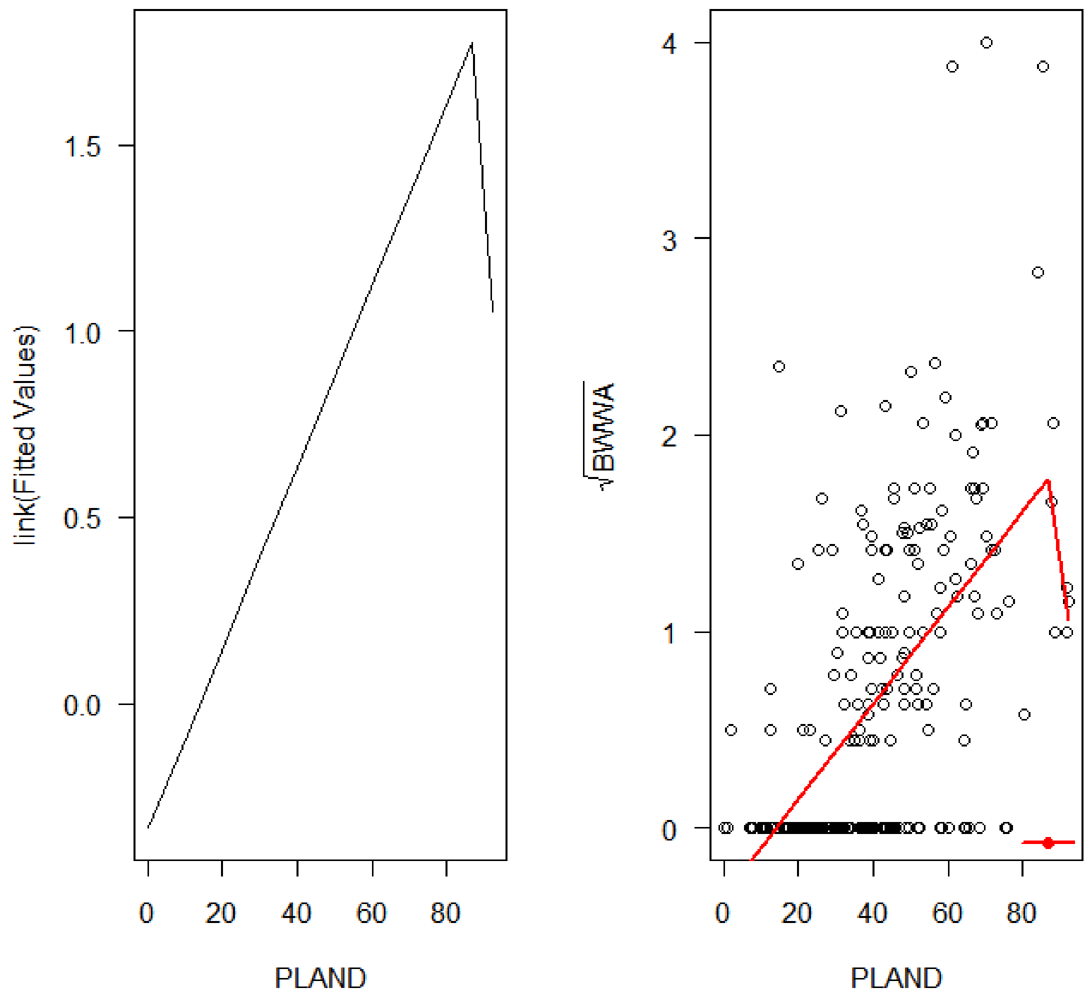

Figure 3.9: The one-breakpoint model in the package segmented for BWWA at the 180 m buffer size. On the right figure, the red lines represent fitted piecewise regression lines and the red dot at the bottom denotes the breakpoint (threshold) with a bar designating the 95% CI. Note that the scale on the X and Y axes is different. The left figure is an enlargement of the piecewise regression lines. The decline of bird abundance over the threshold can be an effect of some low observations when PLAND is above the threshold. If this were real trend, along with WOTH and AMRE, too much forest plays actually negatively to these species.

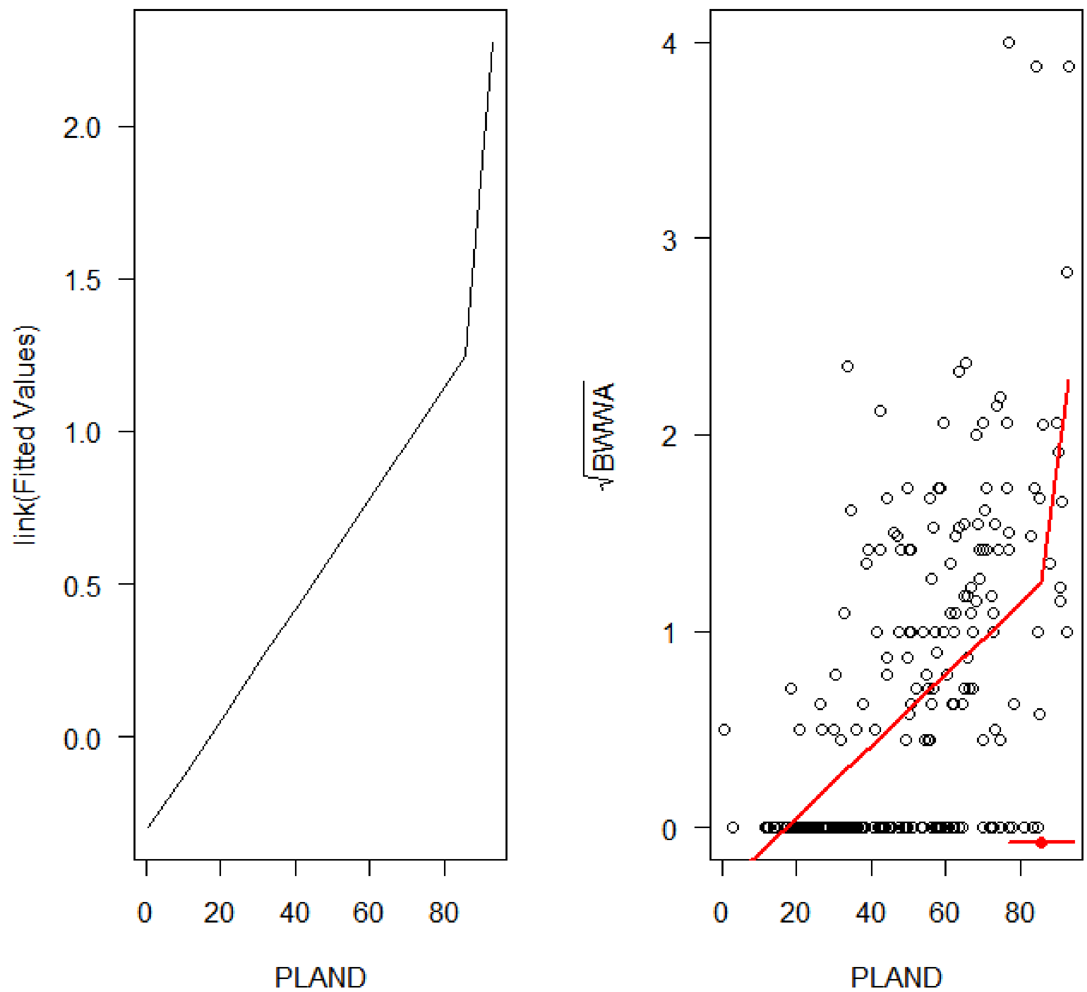

Figure 3.10: The one-breakpoint model in the package segmented for BWWA at the 6 km buffer size. The CI of the slope of the right segment includes 0, which means that this part of the relationship is not stable (could be a flat line).

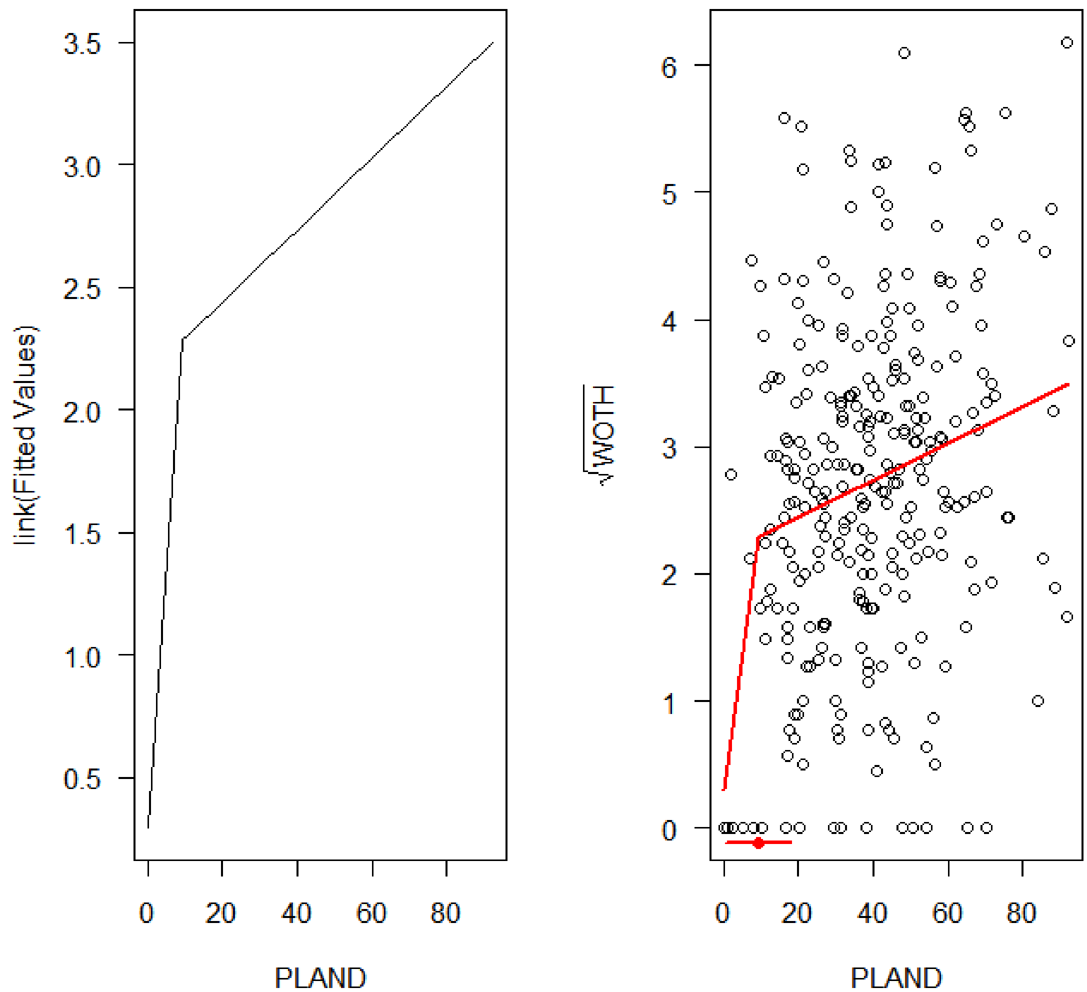

Figure 3.11: The one-breakpoint model in the package segmented for WOTH at the 180 m buffer size. The sharp decline of bird abundance below the threshold may be an artifact of many 0s near low % forest. For the conservation of WOTH, keeping percent forest above the threshold level (9%) seems to be critical to maintain its populations.

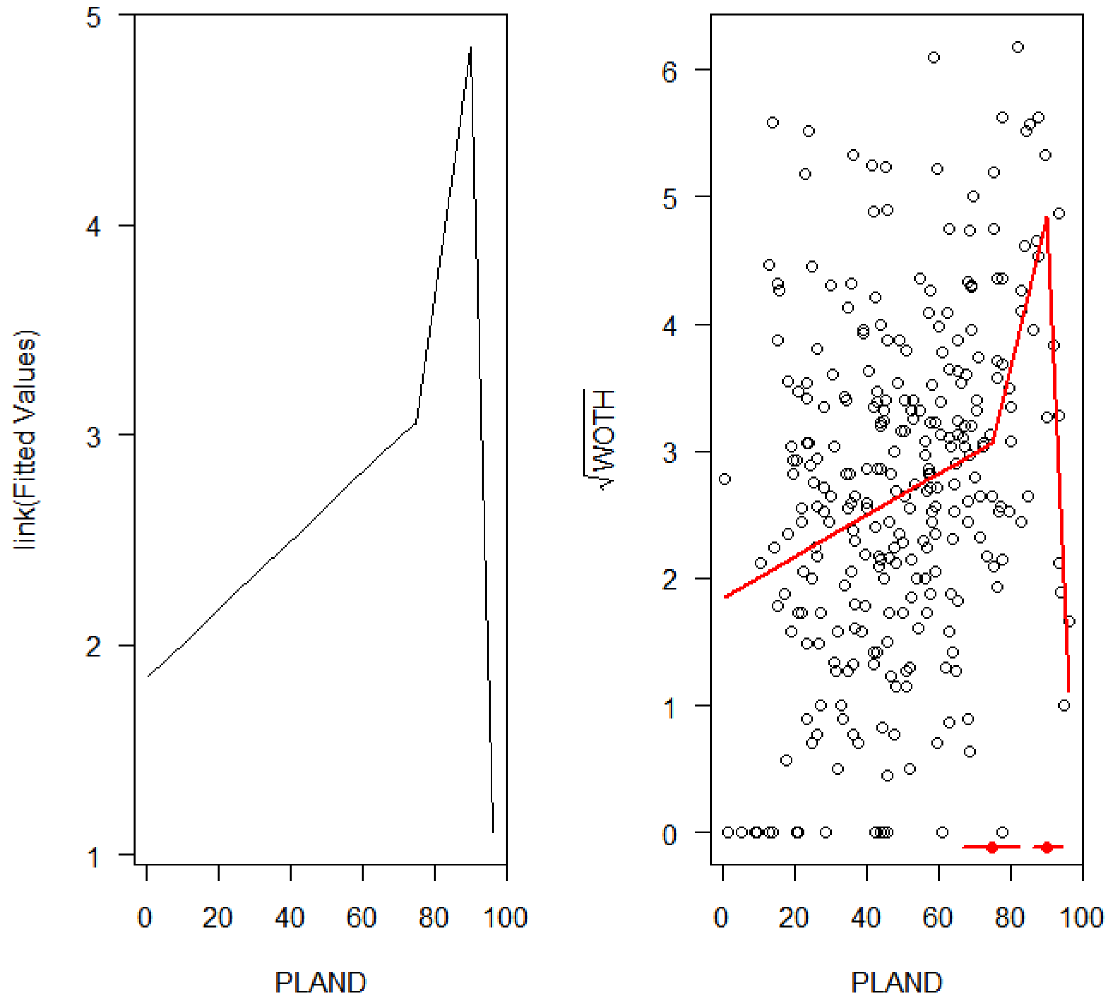

Figure 3.12: The two-breakpoint model in the package segmented for WOTH at the 2010 m buffer size. The decline of bird abundance over the higher threshold can be an artifact of some low observations when PLAND is above the threshold. If this were a real trend, along with AMRE and BWWA, too much forest actually negatively affects these species.

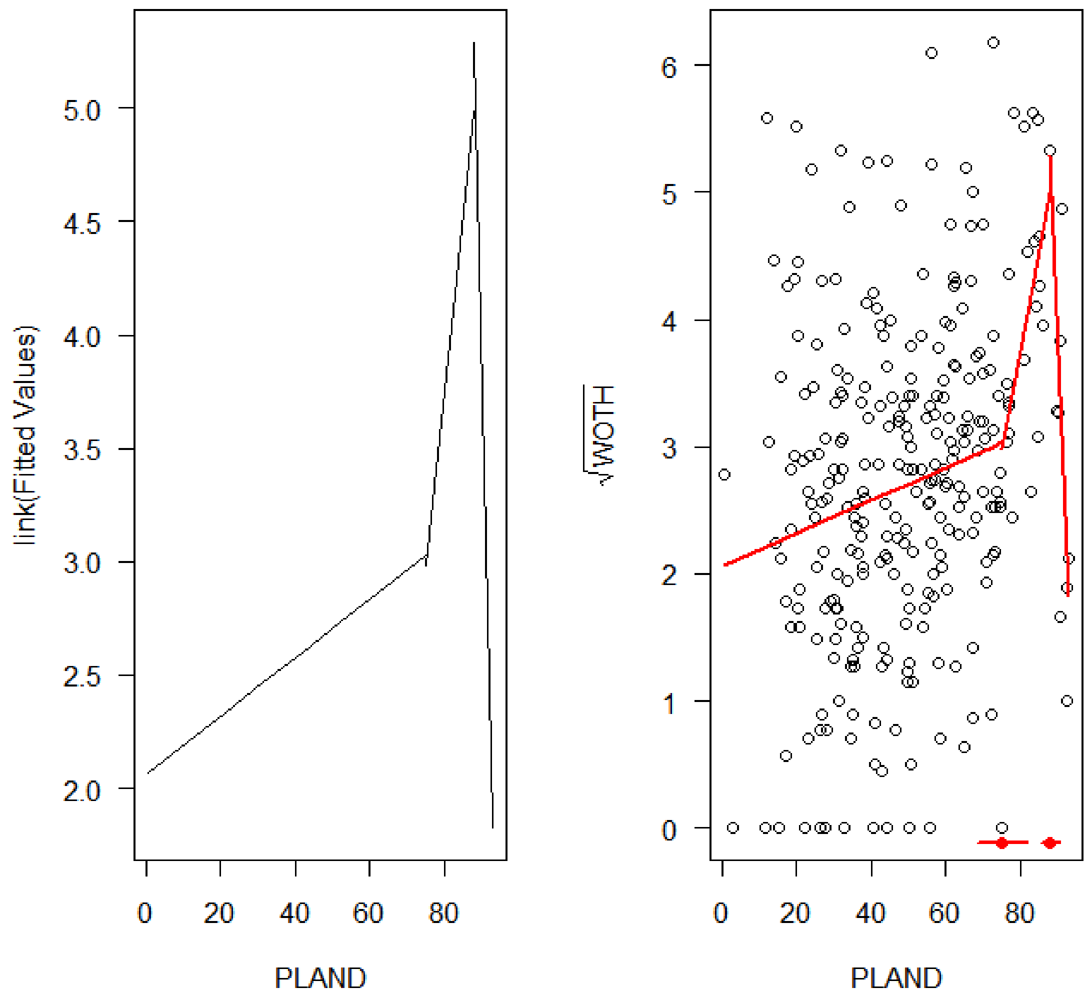

Figure 3.13: The two-breakpoint model in the package segmented for WOTH at the 6 km buffer size.

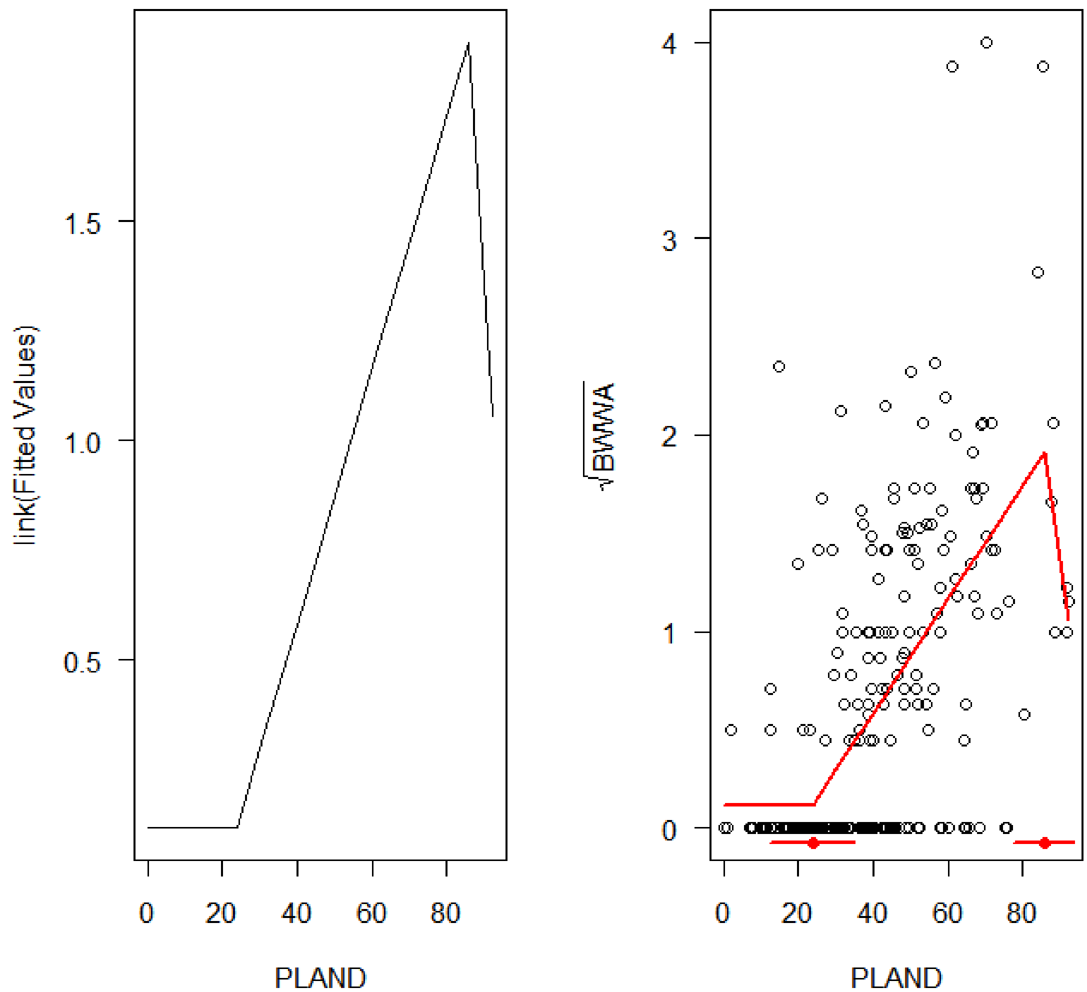

Figure 3.14: The two-breakpoint model in the package segmented for BWWA at the 180 m buffer size. Bird abundance keeps decreasing until 24% threshold. If percent forest is decreased below this level, its effect on bird abundance becomes essentially same as no forest.

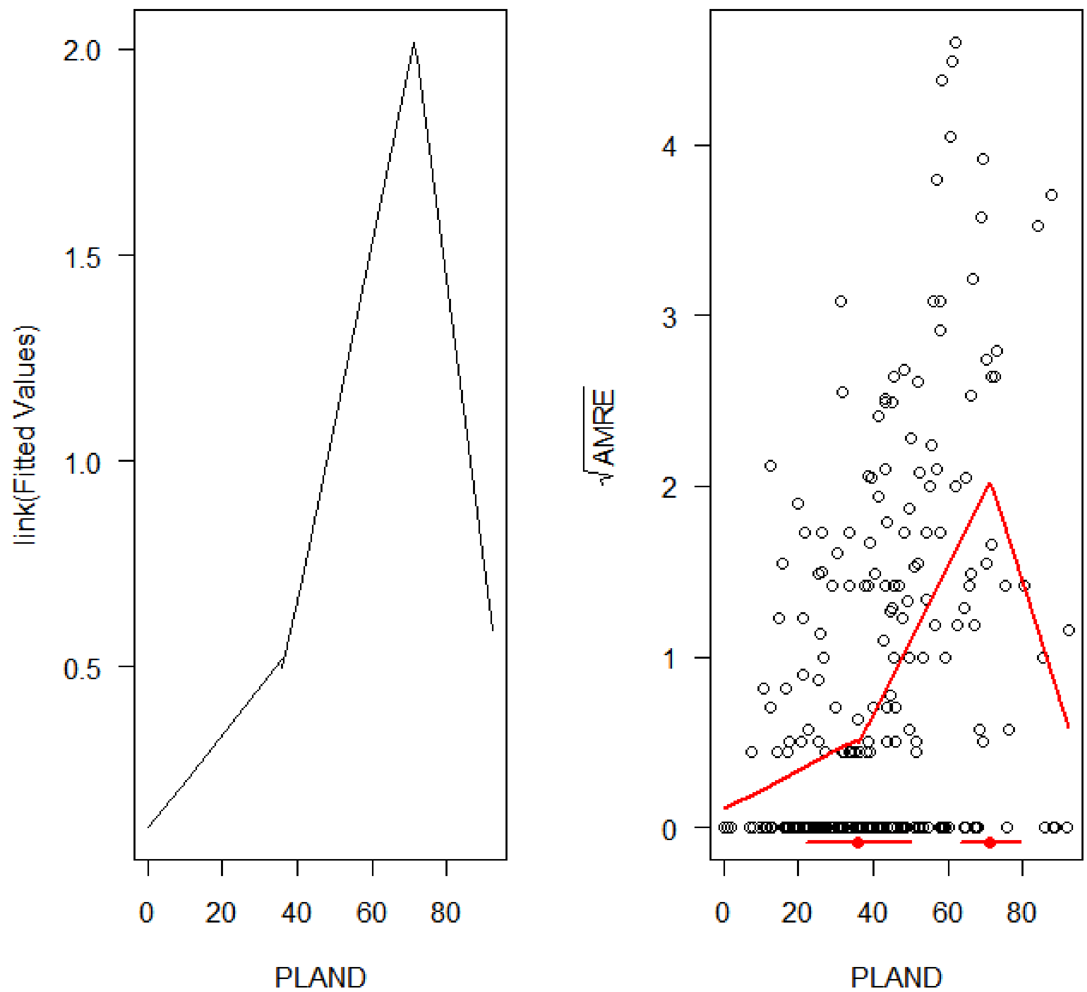

Figure 3.15: The two-breakpoint model in the package segmented for AMRE at the 180 m buffer size. The higher threshold can be an artifact of 0 values when PLAND > 80%. Whether or not this is a real phenomenon needs to be explored further. Overall, the rate of decline of bird abundance is shaper when % forest is high and lower when % forest is low (below 36% threshold). Unit increase in % forest contributes more to the increase in bird abundance over this threshold, and this has a management significance in terms of how much effort should be spent on protecting and restoring forests.

3.4 Discussion

The simple linear regression results show that there is a significant ($\alpha = 0.05$), positive relationship between bird abundance and the percentage of forest cover in an area surrounding a bird survey route. The intercepts of the models suggest that even if the percentage of forest were 0, we would still expect to observe between one and three individuals of EAWP and WOTH (with the CIs not containing 0). Together with much lower R^2 values for EAWP and WOTH, this means that the percentage of forest cover in a landscape is not a good predictor for their abundance. On the other hand, the percentage of forest is a very good predictor for OVEN's and BWWA's abundance. These results are corroborated by the findings that forest cover type, among all land cover types, is best correlated with bird abundance for OVEN, BWWA, and AMRE; whereas, other land cover types such as open space, shrub cover, high impervious cover, and low impervious cover are better correlated with bird abundance for EAWP and WOTH.

The results of the multiple regression analysis reveal that forest composition and configuration and the entire landscape mosaic characteristics at any scale always account for much more variance in OVEN and BWWA than in WOTH, AMRE, and EAWP (Table 3.20). This trend does not change for the variables kept in the reduced models (Table 3.21). The important variables in the reduced models reveal that that across scales, OVEN's and BWWA's abundance increase as (1) the percentage of forest cover increases and as (2) the land cover types of the patches in the neighborhood become more similar (in terms of % canopy) and those similar patches become closer and more contiguous in distribution, and (3) forest edge decreases and/or forest edge contrast decreases (Table 3.22). These landscape structure characteristics suggest a landscape with a high percentage of forest cover with the matrix composed of land cover types that are similar to forest cover, and those similar patches being more aggregated. These landscape structure characteristics account for most variation in bird abundance for OVEN and BWWA that have the highest adjusted R^2 values in the reduced models across scales.

The variables in the multiple regression models, either the full or reduced, always explain the least variation in WOTH's abundance. The abundance of WOTH increases as landscape mosaic diversity decreases and as the amount of forest edge in the buffer increases and/or as the contrast in edges involving forest cover type increases (Table 3.22). CWED (+), SIDI (-), CONTAG (+), CWED (-), and PLAND (+) are identified as important predictors for WOTH, AMRE, and EAWP (Table 3.22), the group that consistently has lower R^2 values than OVEN or BWWA across scales. The abundance of these bird species increases as: (1) contrast-weighted forest edge density increases or decreases; (2) landscape mosaic diversity decreases; (3) the composing patches of the landscape mosaic become more aggregated and less interspersed; and (4) the percentage of forest cover in the buffer increases. Note that SIDI and CONTAG are landscape-level metrics, and these landscape mosaic characteristics as well as the amount and spatial configuration of forest patches are important to predict the abundance of WOTH, AMRE, and EAWP.

When the five species are considered together, across scales the percentage of forest cover in the buffer (+) explains the most variance in bird abundance for more than one species. Also, contrast-weighted edge density, whether affecting bird abundance positively or negatively, is an important characteristic in the landscape across scales. At the 2010 m buffer size, % forest (+), SIMI_AM (+), CWED (+), and CWED (-) have relatively large contribution to explaining the variance in bird abundance for more than one species (Table 3.23). This implies that at this focal scale, more forest birds are observed as the percentage of forest cover increases, the land cover types of the patches become more similar (relative to forest cover type) and those similar patches become closer and more contiguous in distribution, and contrast-weighted forest edge density increases or decreases.

To conserve the selected five woodland-breeding bird species together as a group, the percentage of forest cover in a landscape should be high but some forest edges and/or edge contrast should also be maintained. This recommendation is consistent with the conclusion of the earlier studies that classified WOTH and EAWP as forest interior and edge species (Whitcomb et al. 1981) and OVEN, BWWA, and AMRE as forest interior species (Whitcomb et al. 1981, Fahrig 1999). It is well known that forest fragmentation increases the amount of edge (Wilcox and Murphy 1985, Saunders et al. 1991, Ricklefs and Miller 2000, Cooper and Walters 2002, D'Eon 2002, Fahrig 2002, Noss and Daly 2006). Therefore, the conclusion is also consistent with the earlier classification of WOTH and EAWP having higher tolerance to fragmentation than OVEN, BWWA, and AMRE (Whitcomb et al. 1981). These classifications are, in turn, consistent with my finding that CWED (+) is an important predictor for WOTH and EAWP and CWED (-) for BWWA.

Heteroscedasticity (i.e., a pattern of increasing residuals as the fitted values become larger) was the most common problem in the regression diagnostics. The residuals were, in most cases, normally distributed. There were a couple of outliers and influential points, and they were made sure not to be data entry errors. In total, regression diagnostics for the full and reduced models with transformed bird abundance cast some concern for model assumptions but the problems were judged not to be severe enough to discredit the results of the multiple regression analysis. Generalized linear models (GLMs), which are robust to the violation of model assumptions, can also be used to analyze the data. However, they were not used here because the response variable was not count or proportion data where GLMs are best applied (Dobson and Barnett 2008).

Because most of the thresholds identified by the one-breakpoint model in the package segmented have wide CIs and/or the CIs contain 0, the thresholds are unstable and likely unreliable. This may be partially due to the "noisy" bird survey data, rendering even a weak indication of threshold ecologically significant. The only stable thresholds with narrow CIs found by the one-breakpoint piecewise regression models are: for BWWA, at 87% forest cover at the 180 m buffer size and at 86% forest cover at the 6 km buffer size; for WOTH, at 9% forest cover at the 180 m buffer size. The fact that the adjusted R^2 values of the piecewise regression models producing these thresholds are higher than the R^2 values of the simple linear regression models adds another reason for making these thresholds more reliable. Because the adjusted R^2 value for WOTH is still very low at 0.07 and high impervious cover is a better predictor of its abundance than forest cover, the threshold has little planning and management significance of forest cover for WOTH. Because nearly identical threshold forest cover percentages are identified for two out of the three scales for BWWA, this seems to be a persistent threshold. Moreover, the high threshold value of about 86% is in line with the simple linear regression results that BWWA has the highest or second highest R^2 values across scales. The high threshold is also consistent with BWWA's classification as an area-sensitive species (Whitcomb et al. 1981). However, maintaining an average forest cover in an urban region above this threshold value would be unrealistic. Instead, based on the data analysis conducted, I would recommend protecting large forest patches—for example, large enough to contain an interior area for a breeding territory—and maintaining their connectivity in the urban region because BWWA is a forest interior species (Whitcomb et al. 1981, Fahrig 1999) with very low tolerance to fragmentation (Whitcomb et al. 1981). Forest cover connectivity can be measured by connectivity metrics such as COHESION, CONNECT, GYRTAE_AM, ENN_AM, PROX_AM, and SIMI_AM at the forest cover class level.

3.5 Planning and Management Implications

Based on the results of the study, the following planning and management implications are suggested. First, species specific requirements matter even though all the species in this study are forest-breeding birds and neotropical migrants. To conserve forest birds, species-specific habitat requirements need to be taken into consideration even though the selected species in this study belong to the same guild (woodland breeding and neotropical migrants), sharing similar life history characteristics. For example, for WOTH, which consistently has the lowest R^2 values and adjusted R^2 values for the full and the reduced multiple regression models across scales, the entire landscape mosaic needs to be less diverse and there need to be some forest edges and/or higher edge contrast. For EAWP, which has the lowest R^2 values of the simple linear regression models across scales, other land cover types such as high impervious cover, low impervious cover, and open space can better predict its abundance, and therefore, these cover types need to be in the landscape for its conservation. This conclusions is supported by the fact that CONTAG (+) and CWED (+) were most often selected as important variables in the reduced multiple regression models. In other words, EAWP’s abundance increases as forest edge density increases and/or edge contrast increases by these other land cover types abutting forest cover type, and as patches become more aggregated (i.e., more like-cell adjacencies) and less interspersed (i.e., inequitable distribution of pairwise adjacencies). For EAWP and WOTH, the species better predicted by other land cover types than forest cover, the planning and management of these land cover types in the entire landscape mosaic, such as the land cover diversity and their spatial configurations, is important for the conservation of these bird species. On the other hand, for OVEN, BWWA, and AMRE, the percentage of forest cover in a landscape is the most important factor for their abundance. Therefore, a high percentage of forest cover needs to be maintained in the landscape.

Second, the results of this study are overall consistent with the earlier classifications of the five species. WOTH and EAWP are classified as forest interior and edge species (Whitcomb et al. 1981), having higher tolerance to fragmentation than AMRE, OVEN, or BWWA (Whitcomb et al. 1981), and having small percent forest requirement to be present (Vance et al. 2003). On the other hand, OVEN and BWWA are classified as area-sensitive (Whitcomb et al. 1981, Robbins et al. 1989, Lee et al. 2002), forest interior species (Whitcomb et al. 1981, Fahrig 1999), and having very low tolerance to fragmentation (Whitcomb et al. 1981). For OVEN and BWWA’s conservation, the percentage of forest in the landscape needs to be high and each forest patch needs to be large and well connected to other forest patches. This conclusion is consistent with the variables selected as important in the reduced models for these species: PLAND (+) and SIMI_AM (+) for OVEN and PLAND (+) and CWED (-) for BWWA. My study, however, did not find any evidence to match WOTH's classification as forest area-sensitive species (Robbins et al. 1989, Lee et al. 2002).

Third, for the identified thresholds in the one-breakpoint models, about 86% forest cover for BWWA is too high and 9% for WOTH is too low to be realistic in terms of managing urban forest cover. Moreover, the threshold response for BWWA at the 180 m buffer size indicates a possible negative effect of having too much forest (Figure 3.9). American Forests recommend an average tree canopy cover of 40 percent of the land area for cities east of the Mississippi and in the Pacific Northwest (American Forests 2010). For downtown areas, they recommend 15 percent cover; for urban residential areas, 25 percent cover; and for suburban residential areas, 50 percent cover (American Forests 2010). These percentage values are meant to be general goal guidelines to achieve environmental and quality of life goals, including federal and local clean air and water regulations (American Forests 2010). In the end, each community must set its own tree canopy cover goals (American Forests 2010). As different land uses have varying potential to be forested, parks, residential areas, and vacant lands should be targeted for sustaining or increasing tree cover (Nowak et al. 1996). Nowak et al. (1996) argue that the composition and spatial configuration of land uses in a city largely decide the amount and spatial configuration of tree cover.

Threshold-based planning has an advantage of being a proactive planning, taking actions before the amount of forest in an urban region is reduced below the threshold level or it can serve as a useful target of restoration. This would translates to conservation planning actions such as prioritizing land management or acquisition options, and targeting areas for restoration. The obstacles to threshold-based planning include a lack of species-specific data, difficulty in detecting thresholds, and the danger of over-simplifying complex social-ecological systems.

Overall, in terms of forest bird conservation and management, the percentage of forest cover in the landscape (+) and contrast-weighted edge density, regardless of its sign, are the most important predictors of bird abundance across scales. These variables should be monitored for conservation and management. Edges and/or high edge contrast can be intentionally created, removed, and mitigated by land-use planning. For OVEN, BWWA, and AMRE—those species that are more sensitive to forest loss and fragmentation—foremost, the percentage of forest cover in a landscape needs to be high. Planning and management efforts should focus on protecting as much forest cover as possible and restoring it where possible. For OVEN and BWWA, and at the 2010 m buffer size, similar land cover types (in terms of % canopy) need to be maintained in the matrix and those patches of similar land cover types should be closer and more contiguous in distribution, and more contiguous forest patches with fewer high contrast edges should be maintained. On the other hand, for EAWP and WOTH, the percentage of forest cover is not the most important factor for bird abundance; rather, maintaining some forest edges and/or strong edge contrast is important for the conservation of these more fragmentation-tolerant species. Therefore, it is important not to be too concerned about trying to maintain only large, contiguous forest patches. As for the management of broader landscapes, for EAWP and WOTH, the entire landscape mosaic needs to be less diverse and patches of different land cover types need to be more aggregated and less interspersed as well.

Finally, I emphasize the importance of broad-scale, species-habitat relationship studies. They can contribute to developing a regional goal for biodiversity conservation and to advancing landscape ecological planning that would support biodiversity in a broader urban region.

Table 3.1: Population trend analysis of woodland breeding bird species between 1980 and 2005. The species that showed significant (*p* <0.05) declining trends in the particular state during the period are marked as 1. The percentage is the percentage of 19 states.

Species	AL	CT	FL	GA	KY	ME	MD	MA	MS	NH	NJ	NY	NC	PA	SC	TN	VT	VA	WV	%
Acadian Flycatcher			1										1	1					1	21.1
American Redstart						1				1		1	1				1			26.3
Bachman's Sparrow			1												1					10.5
Barred Owl																1				5.3
Black-and-white Warbler	1					1	1	1		1		1		1		1		1		47.4
Black-billed Cuckoo		1								1	1			1			1	1	1	36.8
Black-capped Chickadee								1												5.3
Black-thr. Blue Warbler												1								5.3
Blue-gray Gnatcatcher														1					1	10.5
Broad-winged Hawk	1								1											10.5
Brown Creeper								1												5.3
Brown-headed Nuthatch	1																			5.3
Canada Warbler						1						1								10.5
Carolina Chickadee	1			1					1				1					1		26.3
Cerulean Warbler																			1	5.3
Chuck-will's-widow	1		1	1	1															21.1
Downy Woodpecker	1												1			1		1		21.1
Eastern Wood-Pewee	1			1	1	1		1		1		1	1	1	1		1	1	1	68.4
Evening Grosbeak						1														5.3
Grt. Crested Flycatcher												1		1					1	15.8
Hairy Woodpecker	1																			5.3
Hooded Warbler			1										1							10.5
Kentucky Warbler	1						1							1				1	1	26.3
Least Flycatcher		1						1		1		1		1			1			31.6
Louisiana Waterthrush		1										1								10.5

Continued on next page

Table 3.1, continued

Species	AL	CT	FL	GA	KY	ME	MD	MA	MS	NH	NJ	NY	NC	PA	SC	TN	VT	VA	WV	%
Northern Parula									1				1		1					15.8
Northern Waterthrush										1										5.3
Ovenbird	1	1						1			1					1				26.3
Pileated Woodpecker	1																			5.3
Prothonotary Warbler	1		1	1											1					21.1
Purple Finch								1		1										10.5
Red-eyed Vireo			1				1	1					1							21.1
Rose-breasted Grosbeak		1					1	1		1	1	1		1						36.8
Ruby-thr. Hummingbird									1											5.3
Ruffed Grouse												1								5.3
Scarlet Tanager							1			1		1		1						21.1
Summer Tanager	1														1					10.5
Swainson's Warbler													1							5.3
Tennessee Warbler						1														5.3
Veery						1		1		1	1	1					1			31.6
Warbling Vireo																			1	5.3
Whip-poor-will					1		1													10.5
Wood Thrush			1	1	1	1	1	1		1	1	1	1	1	1		1	1	1	78.9
Worm-eating Warbler														1						5.3
Yellow-billed Cuckoo	1	1		1			1		1		1		1			1				42.1

Table 3.2: Population trend analysis of open-cup nesting bird species between 1980 and 2005. The species that showed significant (p <0.05) declining trends in the particular state during the period are marked as 1. The percentage is the percentage of 19 states.

Species	AL	CT	FL	GA	KY	ME	MD	MA	MS	NH	NJ	NY	NC	PA	SC	TN	VT	VA	WV	%
Acadian Flycatcher			1										1	1					1	21.1
American Goldfinch																			1	5.3
American Redstart						1				1		1	1				1			26.3
American Robin	1							1						1						15.8
Bachman's Sparrow			1												1					10.5
Barn Swallow						1		1		1		1		1		1	1	1	1	47.4
Black-and-white Warbler	1					1	1	1				1		1		1		1		42.1
Black-billed Cuckoo		1								1	1			1			1	1	1	36.8
Black-thr. Blue Warbler												1								5.3
Blue Grosbeak																		1		5.3
Blue Jay	1	1	1	1	1			1		1	1		1			1		1		57.9
Blue-gray Gnatcatcher														1					1	10.5
Blue-winged Warbler		1									1	1							1	21.1
Bobolink						1											1			10.5
Brown Thrasher		1			1			1		1	1	1				1				36.8
Canada Warbler						1						1								10.5
Cedar Waxwing								1									1			10.5
Cerulean Warbler																			1	5.3
Chestnut-sided Warbler						1		1		1							1			21.1
Chipping Sparrow												1		1					1	15.8
Common Grackle	1		1	1	1		1	1			1	1	1	1	1	1	1	1	1	78.9
Common Yellowthroat		1	1		1	1		1	1	1	1	1					1		1	57.9
Eastern Kingbird		1	1		1	1	1	1		1	1	1				1				52.6

Continued on next page

Table 3.2, continued

Species	AL	CT	FL	GA	KY	ME	MD	MA	MS	NH	NJ	NY	NC	PA	SC	TN	VT	VA	WV	%
Eastern Meadowlark	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	100.0
Eastern Phoebe																	1			5.3
Eastern Towhee		1	1	1		1		1		1	1	1								42.1
Eastern Wood-Pewee	1			1	1	1		1				1	1	1	1		1	1	1	63.2
Evening Grosbeak						1														5.3
Field Sparrow	1	1		1	1	1	1	1		1	1	1		1				1	1	68.4
Golden-winged Warbler																			1	5.3
Grasshopper Sparrow				1			1					1		1				1	1	31.6
Gray Catbird					1	1				1					1	1		1		31.6
Hooded Warbler			1										1							10.5
Indigo Bunting			1		1			1		1	1	1	1	1				1	1	52.6
Kentucky Warbler	1						1							1				1	1	26.3
Least Flycatcher		1				1		1		1		1		1			1			36.8
Loggerhead Shrike	1		1		1								1			1				26.3
Louisiana Waterthrush		1										1								10.5
Nashville Warbler						1											1			10.5
Northern Mockingbird	1	1	1	1			1	1		1	1									42.1
Northern Parula									1				1		1					15.8
Northern Waterthrush										1										5.3
Prairie Warbler		1		1	1		1				1					1			1	36.8
Purple Finch								1		1										10.5
Red-eyed Vireo			1				1	1					1							21.1
Red-winged Blackbird	1	1	1	1					1			1		1	1	1				47.4
Rose-breasted Grosbeak		1					1	1		1	1	1		1						36.8
Savannah Sparrow												1		1						10.5
Scarlet Tanager							1			1		1		1						21.1

Continued on next page

Table 3.2, continued

Species	AL	CT	FL	GA	KY	ME	MD	MA	MS	NH	NJ	NY	NC	PA	SC	TN	VT	VA	WV	%
Song Sparrow		1						1		1	1	1		1			1	1		42.1
Summer Tanager	1														1					10.5
Swainson's Warbler													1							5.3
Tennessee Warbler						1														5.3
Veery						1		1		1	1	1					1			31.6
Vesper Sparrow														1						5.3
Warbling Vireo																			1	5.3
White-eyed Vireo							1				1				1				1	21.1
White-throated Sparrow						1				1							1			15.8
Wood Thrush			1	1	1	1	1	1		1	1	1	1	1	1		1	1	1	78.9
Worm-eating Warbler														1						5.3
Yellow Warbler					1	1				1		1				1	1	1		36.8
Yellow-billed Cuckoo	1	1		1			1		1		1		1			1				42.1
Yellow-breasted Chat			1				1							1					1	21.1

Table 3.3: Reclassified land cover classes. The original NLCD 2001 land cover classes are aggregated to seven functional land cover classes (the right most column).

NLCD 2001 Class Code	NLCD 2001 Description	Modified Class Code	Modified Description	Functional Class Code	Functional Land Cover Classification
11	Open Water	1	Water	1	Unvegetated
12	Perennial Ice/Snow	9	Ice/Snow	1	
21	Developed, Open Space**	2	Urban	2	Open Space
22	Developed, Low Intensity	2		3	Low Imperviousness
23	Developed, Medium Intensity	2		4	High Imperviousness
24	Developed, High Intensity	2		4	
31	Barren Land, Rock, Sand, Clay	3	Barren	1	
32	Unconsolidated Shore*	3		1	
41	Deciduous Forest	4	Forest	5	Forest
42	Evergreen Forest	4		5	
43	Mixed Forest	4		5	
52	Shrub/Scrub	5	Shrub	6	Shrub
71	Grassland/Herbaceous	6	Herbaceous	7	Herbaceous
81	Pasture, Hay	7	Agriculture	7	
82	Cultivated Crops#	7		7	
90	Woody Wetlands##	8	Wetlands	6	
91	Palustrine Forested Wetland*	8		5	
92	Palustrine Scrub/Shrub Wetland*	8		6	
93	Estuarine Forested Wetland*	8		5	
94	Estuarine Scrub/Shrub Wetland*	8		6	
95	Emergent Herbaceous Wetlands	8		7	
96	Palustrine Emergent Wetland*	8		7	
97	Estuarine Emergent Wetland*	8		7	
98	Palustrine Aquatic Bed*	8		7	
99	Estuarine Aquatic Bed*	8		7	

* Coastal Areas Only

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Table 3.3 Notes, continued

** Developed, Open Space has < 20% impervious surfaces, mostly lawn but some planted trees in parks, golf courses, yards, and for recreation, erosion control, and aesthetic purposes
# Cultivated crops do include perennial woody crops such as orchards and vineyards
Woody Wetlands are included in "Shrub" due to their lower potential use as breeding habitat by the selected forest bird species

Table 3.4: Definitions of landscape metrics in this study (adopted from McGarigal et al. 2002).

Landscape metrics	Abbreviation	Description	Units	Range
Class-level				
Percentage of landscape	PLAND	PLAND equals the sum of the areas of all patches of the corresponding patch type (i.e., reclassified land cover class) divided by total buffer area, multiplied by 100 (to convert to a percentage).	Percent	0 < PLAND <= 100
Patch density	PD	PD equals the number of forest patches divided by total buffer area, multiplied by 10,000 and 100 (to convert to 100 ha).	Number per 100 hectares	0 <, constrained by cell size
Radius of gyration (area-weighted mean)	GYRATE_AM	Also known as correlation length, GYRATE_AM is average extensiveness of connected cells. GYRATE_AM is computed as the area-weighted mean radius of gyration across all forest patches in a certain buffer size. GYRATE_AM is another measure of connectedness.	Meters	0 =<, without limit
Shape index (area-weighted mean)	SHAPE_AM	SHAPE equals patch perimeter (given in number of cell surfaces) divided by the minimum perimeter (given in number of cell surfaces) possible for a maximally compact patch of the corresponding patch area. SHAPE_AM is the area-weighted mean of SHAPE across all forest patches.	None	1 <=, without limit

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Table 3.4, continued

Fractal dimension index (area-weighted mean)	FRAC_AM	FRAC equals 2 times the logarithm of patch perimeter divided by the logarithm of patch area. FRAC_AM is the area-weighted mean of FRAC across all forest patches.	None	1 =< FRAC <= 2
Perimeter-area fractal dimension	PAFRAC	PAFRAC equals 2 divided by the slope of regression line obtained by regressing the logarithm of patch area against the logarithm of patch perimeter.	None	1 =< PAFRAC <= 2
Proximity index (area-weighted mean)	PROX_AM	PROX equals the sum of patch area divided by the nearest edge-to-edge distance squared between the patch and the focal patch of all patches of the corresponding patch type whose edges are within a specified distance of the focal patch. PROX_AM is the area-weighted mean of PROX across all forest patches.	None	0 =<
Similarity index (area-weighted mean)	SIMI_AM	SIMI equals the sum, over all neighboring patches with edges within a specified distance of the focal patch, of neighboring patch area times a similarity coefficient between the focal patch type and the class of the neighboring patch, divided by the nearest edge-to-edge distance squared between the focal patch and the neighboring patch. SIMI_AM is the area-weighted mean of SIMI across all forest patches.	None	0 =<
Euclidean nearest-neighbor distance (area-weighted mean)	ENN_AM	ENN is a measure of isolation. ENN equals the distance from a forest patch to the nearest neighboring forest patch, based on shortest edge-to-edge distance. The edge-to-edge distances are from cell center to cell center. ENN_AM is the area-weighted mean of ENN across all forest patches.	Meters	60 =<, without limit
Contrast-weighted edge density	CWED	CWED equals the sum of the lengths of each forest edge segment multiplied by the corresponding contrast weight, divided by the total buffer area, multiplied by 10,000 (to convert to hectares).	Meters per hectare	0 =<, without limit

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Table 3.4, continued

Patch cohesion index	COHESION	COHESION equals 1 minus the sum of patch perimeter divided by the sum of patch perimeter times the square root of patch area for forest patches, divided by 1 minus 1 over the square root of the total number of cells in the buffer area, multiplied by 100 (to convert to a percentage). In short, COHESION measures physical connectedness of forest cover type.	Percent	0 =< COHESION < 100
Connectance index	CONNECT	CONNECT equals the number of functional joinings between all patches of forest cover type, divided by the total number of possible joinings between all patches of forest cover type, multiplied by 100 to convert to a percentage. CONNECT is a common connectivity metric.	Percent	0 =< CONNECT <= 100
Landscape-level				
Simpson's diversity index	SIDI	SIDI equals 1 minus the sum, across all patch types, of the proportional abundance of each patch type squared. It represents the diversity (number and evenness) of the reclassified land cover types in the buffer.	None	0 =< SIDI < 1
Contagion	CONTAG	CONTAG is the probability that two randomly chosen adjacent cells belong to the same land cover class. CONTAG equals 1 minus the sum of the proportional abundance of each patch type multiplied by the proportion of adjacencies between cells of that patch type and another patch type, multiplied by the logarithm of the same quantity, summed over each unique adjacency type and each patch type; divided by 2 times the logarithm of the number of patch types; multiplied by 100 (to convert to a percentage).	Percent	0 < CONTAG <= 100

Table 3.5: Average percent canopy for land cover types. The values are used to calculate similarity coefficients and contrast weights.

Land cover classes	Average % canopy
Unvegetated	0.43
Open Space	25.37
Low Imperviousness	9.10
High Imperviousness	1.28
Forest	80.19
Shrub	64.41
Herbaceous	1.99

Table 3.6: Simple linear regression result at the 180 m buffer size.

	AMRE	BWWA	OVEN	EAWP	WOTH
R^2	0.1696	0.323	0.3946	0.04977	0.06351
Model	y = 0.0224x - 0.0961	y = 0.0228x - 0.2883	y = 0.0551x - 0.0028	y = 0.0123x + 1.4469	y = 0.0176x + 2.0030
F-statistic	51.27	115.9	156.4	14.98	19.33
p-value	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
95% CI of the slope estimate	(0.0162, 0.0285)	(0.0186, 0.0269)	(0.0464, 0.0637)	(0.0061, 0.0186)	(0.0097, 0.02554)
95% CI of the intercept estimate	(-0.3703, 0.1781)	(-0.4732, -0.1034)	(-0.3889, 0.3832)	(1.1724, 1.7213)	(1.6557, 2.3503)

Table 3.7: Multiple regression result at the 180 m buffer size, showing R^2 values for the full model and adjusted R^2 values for the reduced model.

Species	Full Model	Subset Model	Number of Variables	Variables in the Subset	More Important Variables***
EAWP	0.2951	0.2653	6	PD, PROX_AM*, SIMI_AM*, ENN_AM*, CWED*, CONTAG*	CONTAG (-)**, CWED (-), PROX_AM (-), SIMI_AM (+)
WOTH	0.2260	0.1487	3	CWED*, COHESION*, SIDI*	COHESION (+), SIDI (-), CWED (+)
AMRE	0.3147	0.2299	3	FRAC_AM*, PAFRAC*, PROX_AM*	FRAC_AM (+)
BWWA	0.3983	0.3609	4	PLAND*, PROX_AM*, SIMI_AM*, CWED	PLAND (+)
OVEN	0.4612	0.4428	5	PLAND*, PAFRAC, PROX_AM*, SIMI_AM, CWED*	PLAND (+)

* The variable is statistically significant at $\alpha < 0.05$

** The sign in the parenthesis is the sign of the variable's coefficient

*** More important variables are listed in the order of relative importance in explaining the total variance of bird abundance in the reduced model

Table 3.8: Threshold detection, segmented package result, at the 180 m buffer size.

One-breakpoint Model					
	EAWP	WOTH	AMRE	BWWA	OVEN
Breakpoint estimate	8.235	9.356	31.01	86.69	8.36
CI	(-0.437, 16.91)	(0.6091, 18.1)	(-124.3, 186.3)	(79.86, 93.53)	(-67.03, 83.75)
Adjusted R2	0.05809	0.07406	0.1615	0.3265	0.387
Two-breakpoint Model					
	EAWP	WOTH	AMRE	BWWA	OVEN
Breakpoint estimates	N/A	N/A	36.12, 71.39	23.86, 85.67	N/A
CIs	N/A	N/A	(22.12, 50.12), (63.56, 79.22)	(12.75, 34.97), (77.51, 93.82)	N/A
Adjusted R2	N/A	N/A	0.2171	0.3354	N/A

Table 3.9: R^2 values of the linear regression between bird abundance and each land cover type at the 180 m buffer size.

Species	Unvegetated	Open Space	Low Imperviousness	High Imperviousness	Forest	Shrub	Herbaceous
EAWP	0.04191*	0.1237*	0.146*	0.1723*	0.04977*	0.01967*	0.04298*
WOTH	0.05796*	0.04687*	0.04733*	0.1197*	0.06351*	0.04906*	0.00738
AMRE	0.00779	0.00090	0.02237*	0.02335*	0.1696*	0.00832	0.1147*
BWWA	0.00060	0.01203	0.05133*	0.03571*	0.323*	0.01365	0.1925*
OVEN	0.00667	0.0866*	0.07308*	0.06158*	0.3946*	0.02524*	0.1644*

* p-value < 0.05

Table 3.10: Simple linear regression result at the 2010 m buffer size.

	AMRE	BWWA	OVEN	EAWP	WOTH
R^2	0.1913	0.3161	0.329	0.05766	0.100
Model	y = 0.0215x - 0.2856	y = 0.0203x - 0.4007	y = 0.0455x - 0.0954	y = 0.0119x + 1.3488	y = 0.0200x + 1.7150
F-statistic	59.37	112.3	117.7	17.5	31.67
p-value	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
95% CI of the slope estimate	(0.0160, 0.0270)	(0.0166, 0.02411)	(0.0373, 0.0538)	(0.0063, 0.0176)	(0.0130, 0.0270)
95% CI of the intercept estimate	(-0.5874, 0.0161)	(-0.6071, -0.1942)	(-0.5487, 0.3579)	(1.0495, 1.6482)	(1.3395, 2.0904)

Table 3.11: Multiple regression result at the 2010 m buffer size, showing R^2 values for the full model and adjusted R^2 values for the reduced model.

Species	Full Model	Subset Model	Number of Variables	Variables in the Subset	More Important Variables***
EAWP	0.2855	0.2524	7	GYRATE_AM*, FRAC_AM*, PAFRAC*, CWED*, COHESION, CONNECT*, CONTAG*	CONTAG (+)**, CWED (+), GYRTAE_AM (+)
WOTH	0.2525	0.1954	2	SIDI*, CWED*	SIDI (-), CWED (+)
AMRE	0.2771	0.2278	4	PLAND*, SHAPE_AM*, CWED*, PAFRAC*,	PLAND (+), CWED (-)
BWWA	0.4148	0.3948	5	PLAND*, SIMI_AM*, CWED*, PAFRAC, SHAPE_AM	PLAND (+), SIMI_AM (+), CWED (-)
OVEN	0.4360	0.4066	4	PLAND*, SIMI_AM*, CWED*, PAFRAC*	PLAND (+), SIMI_AM (+)

* The variable is statistically significant at $\alpha < 0.05$

** The sign in the parenthesis is the sign of the variable's coefficient

*** More important variables are listed in the order of relative importance in explaining the total variance of bird abundance in the reduced model

Table 3.12: Threshold detection, segmented package result, at the 2010 m buffer size.

One-breakpoint Model					
	EAWP	WOTH	AMRE	BWWA	OVEN
Breakpoint estimate	52.63	54.19	51.34	31.13	2.125
CI	(14.37, 90.9)	(17.3, 91.08)	(25.24, 77.45)	(15.72, 46.54)	(-1703, 1708)
Adjusted R2	0.05168	0.09503	0.1913	0.3224	0.3205
Two-breakpoint Model					
	EAWP	WOTH	AMRE	BWWA	OVEN
Breakpoint estimates	N/A	74.37, 90.13	N/A	21.91, 60.59	46.82, 48.56
CIs	N/A	(66.58, 82.16), (86.22, 94.03)	N/A	(-3.481, 47.30), (39.57, 81.61)	(43.28, 50.36), (40.05, 57.07)
Adjusted R2	N/A	0.1328	N/A	0.3226	0.3334

Table 3.13: R^2 values of the linear regression between bird abundance and each land cover type at the 2010 m buffer size.

Species	Unvegetated	Open Space	Low Imperviousness	High Imperviousness	Forest	Shrub	Herbaceous
EAWP	0.06823*	0.1392*	0.1263*	0.09241*	0.05766*	0.04185*	0.03476*
WOTH	0.08935*	0.03012*	0.02896*	0.03682*	0.1*	0.1011*	0.00381
AMRE	0.00906	0.01275	0.03916*	0.02601*	0.1913*	0.00002	0.1278*
BWWA	0.00519	0.05764*	0.07555*	0.04154*	0.3161*	0.01338	0.2344*
OVEN	0.01803*	0.1185*	0.08882*	0.05402*	0.329*	0.02233*	0.1971*

* p-value < 0.05

Table 3.14: Simple linear regression result at the 6 km buffer size.

	AMRE	BWWA	OVEN	EAWP	WOTH
R^2	0.2019	0.2939	0.27	0.0332	0.0703
Model	y = 0.0230x - 0.3617	y = 0.0204x - 0.3982	y = 0.0428x - 0.0667	y = 0.0093x + 1.5168	y = 0.0171x + 1.9087
F-statistic	62.23	99.06	86.92	9.581	21.02
p-value	< 0.001	< 0.001	< 0.001	0.002	< 0.001
95% CI of the slope estimate	(0.0173, 0.0288)	(0.0164, 0.0245)	(0.0338, 0.0519)	(0.0034, 0.0152)	(0.0098, 0.0245)
95% CI of the intercept estimate	(-0.6751, -0.0484)	(-0.6179, -0.1786)	(-0.4271, 0.5605)	(1.2042, 1.8294)	(1.5163, 2.3012)

Table 3.15: Multiple regression result at the 6 km buffer size, showing R^2 values for the full model and adjusted R^2 values for the reduced model.

Species	Full Model	Subset Model	Number of Variables	Variables in the Subset	More Important Variables***
EAWP	0.2459	0.2099	4	PAFRAC*, PROX_AM*, CWED*, CONTAG*	CONTAG (+)**, CWED (+)
WOTH	0.2181	0.1713	3	SIMI_AM, CWED*, SIDI*	SIDI (-)
AMRE	0.3273	0.2775	5	PLAND*, PD*, FRAC_AM*, PAFRAC*, COHESION*	PLAND (+), PD (-)
OVEN	0.4002	0.3404	5	PLAND*, PAFRAC, SIMI_AM*, CONNECT*, SIDI	PLAND (+),SIMI_AM (+), CONNECT (+),
BWWA	0.3723	0.3406	4	PLAND*, PAFRAC*, CWED*, SHAPE_AM	PLAND (+), CWED (-)

* The variable is statistically significant at $\alpha < 0.05$

** The sign in the parenthesis is the sign of the variable's coefficient

*** More important variables are listed in the order of relative importance in explaining the total variance of bird abundance in the reduced model

Table 3.16: Threshold detection, segmented package result, at the 6 km buffer size.

One-breakpoint Model					
	EAWP	WOTH	AMRE	BWWA	OVEN
Breakpoint estimate	49.21	42.54	34.08	85.67	N/A
CI	(13.05, 85.36)	(23.22, 61.86)	(13.07, 55.09)	(76.84, 94.5)	N/A
Adjusted R2	0.0274	0.0728	0.1996	0.3068	N/A
Two-breakpoint Model					
	EAWP	WOTH	AMRE	BWWA	OVEN
Breakpoint estimates	67.97, 79.57	75.16, 87.92	N/A	22.05, 86.00	75.97, 79.48
CIs	(48.22, 87.72), (72.44, 86.70)	(68.39, 81.93), (85.39, 90.45)	N/A	(5.07, 39.04), (77.53, 94.46)	(73.40, 78.55), (75.37, 83.59)
Adjusted R2	0.0588	0.1103	N/A	0.3057	0.106

Table 3.17: R^2 values of the linear regression between bird abundance and each land cover type at the 6 km buffer size.

Species	Unvegetated	Open Space	Low Imperviousness	High Imperviousness	Forest	Shrub	Herbaceous
EAWP	0.02946*	0.1137*	0.1035*	0.09811*	0.0332*	0.05151*	0.042*
WOTH	0.0282*	0.01735*	0.01677*	0.03636*	0.0703*	0.126*	0.00336
AMRE	0.0306*	0.01008	0.04411*	0.0332*	0.2019*	0.00099	0.1267*
BWWA	0.00433	0.05964*	0.08399*	0.03975*	0.2939*	0.0158	0.2267*
OVEN	0.00326	0.1199*	0.09219*	0.0483*	0.27*	0.03158*	0.1827*

* p-value < 0.05

Table 3.18: Classification of the selected species by the earlier studies.

Species	Tolerance to fragmentation (Whitcomb et al. 1981)	Area-sensitive? (Whitcomb et al. 1981, Robbins et al. 1989, Lee et al. 2002)	Classification based on habitat associations (Whitcomb et al. 1981)	Minimum habitat amount (% forest)* (Vance et al. 2003)
EAWP	High (2/max 8)	Not mentioned	Forest interior and edge	1
WOTH	Medium (3/8)	Not mentioned, Yes, Yes	Forest interior and edge	10
BWWA	Very low (6.5/8)	Yes	Forest interior bird (Fahrig 1999 also classifies this species as forest-interior species)	Species not included in the study
OVEN	Very low (6)	Yes, Yes, Yes	Forest interior bird	38
AMRE	Low (5)	Not mentioned	Forest interior bird	62.5

* Minimum habitat (forest) requirements at which there was a 50% probability of presence of the bird species in the landscape over a 10-year period

Table 3.19: Simple linear regression model result comparison across scales. Within the same buffer size, the species are ordered in the increasing R^2 value from the top to the bottom.

	180 m	2010 m	6 km
R^2 lowest	EAWP	EAWP	EAWP
	WOTH	WOTH	WOTH
	AMRE	AMRE	AMRE
	BWWA	BWWA	OVEN
R^2 highest	OVEN	OVEN	BWWA

Table 3.20: Additive, full multiple regression model result comparison across scales. Within the same buffer size, the species are ordered in the increasing R^2 value from the top to the bottom.

	180 m	2010 m	6 km
R^2 lowest	WOTH	WOTH	WOTH
	EAWP	AMRE	EAWP
	AMRE	EAWP	AMRE
	BWWA	BWWA	BWWA
R^2 highest	OVEN	OVEN	OVEN

Table 3.21: Reduced multiple regression model* result comparison across scales. Within the same buffer size, the species are ordered in the increasing adjusted R^2 value from the top to the bottom.

	180 m	2010 m	6 km
Adjusted R^2 lowest	WOTH	WOTH	WOTH
	AMRE	AMRE	EAWP
	EWAP	EAWP	AMRE
	BWWA	BWWA	OVEN
Adjusted R^2 highest	OVEN	OVEN	BWWA

* Percent forest is not necessarily included.

Table 3.22: Important variables in the reduced multiple regression models across scales

Species	Important landscape metrics
WOTH	SIDI (-)*, CWED (+)
EAWP	CONTAG (+), CWED (+)
AMRE	% forest (+)
OVEN	% forest (+)*, SIMI_AM (+)
BWWA	% forest (+)*, CWED (-)
OVEN and BWWA	% forest (+), SIMI_AM (+), CWED (-)
EAWP and AMRE	% forest (+),CONTAG (+), CWED (+), CWED (-)
WOTH, AMRE, and EAWP	CWED (+), SIDI (-), CONTAG (+), CWED (-), % forest (+)

* Identified as important at all three scales.

The other variables listed are identified as important either at two scales or by two species.

Table 3.23: Important variables in the reduced multiple regression models within each buffer size across species. The variables in the first row are identified as important for more than one species.

	180 m	2010 m	6 km
Five species together	% forest (+)	% forest (+), SIMI_AM (+), CWED (+), CWED (-)	% forest (+)
	CWED (+), CWED (-)		CWED (+), CWED (-)

CHAPTER 4

DEVELOPMENT OF A LANDSCAPE PLANNING META-MODEL AND ITS APPLICATION TO GREENSPACE CONSERVATION PLANNING IN URBAN REGIONS BASED ON THE RESILIENCE CONCEPT

4.1 Introduction

4.1.1 Background

4.1.1.1 Biodiversity

The loss of biodiversity is a global concern (Convention on Biological Diversity [CBD] 1992, UNEP 2007, Conservation International 2009, International Union for Conservation of Nature [IUCN] 2010) since, aside from ethical arguments, biodiversity is tied closely to ecosystem services and human well-being (McNeeley et al. 1990, Peck 1998, MA 2005, Groom et al. 2006). The Convention on Biological Diversity (CBD) defines biodiversity as “the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems” (CBD 1992). Biodiversity encompasses multiple levels of biological organization (Noss 1990, Peck 1998, Dale 2001, Groom et al. 2006). Noss (1990) expanded on the three primary attributes of biodiversity recognized by Franklin et al. (1981)—composition, structure, and function—into a nested hierarchy that incorporates elements of each attribute at four levels of organization: regional landscape, ecosystem-community, species-population, and genetic. Biodiversity, therefore, exists and needs to be understood at multiple scales (Ahern et al. 2006).

Acknowledging the wide array of what biodiversity contains and its hierarchical organization, in this chapter biodiversity is dealt with at the level of species, specifically, forest bird species (see chapter 3). Biodiversity can be expressed in terms of the four levels of organization: landscape diversity, ecosystem or community diversity, species diversity, and genetic diversity (Peck 1998). Species diversity refers to the variety of species in a prescribed area (Dale 2001). Species richness (i.e., the number of species in a prescribed area) and abundance (i.e., the number of individuals of a species in a prescribed area or population) are often used as measures of species diversity and are commonly-used indicators of change in the environment (Spray and McGlothlin 2003, Groom et al. 2006, Primack 2008). I will use forest birds as an example of species-level biodiversity in the context of conservation of forest bird species at an urban regional scale because forest birds are often-used indicators of the amount and spatial distribution of forest land cover (Flather and Sauer 1996, Bolger et al., 1997, Mörtberg and Wallentinus 2000, Boulinier et al. 2001, Marzluff 2001, Donovan and Flather 2002, Fernández-Juricic 2004, Hashimoto et al. 2005, Sandström et al. 2006, Pidgeon et al. 2007).

4.1.1.2 Urban Region

Biodiversity conservation should arguably be one of the major goals in urban regional planning. Urban region is defined as a spatial/geographical entity that is composed of interacting abiotic, biotic, and cultural resources, and can be composed of multiple jurisdictions (McDonnell and Pickett 1990, Forman 1995, 2008, Medley et al. 1995, Foresman et al. 1997, Steiner 2002a). Its boundary is determined by some measure of the intensity of urbanization, or human influences on ecosystems in the landscape (McDonnell and Pickett 1990, Forman 1995, Medley et al. 1995, Foresman et al. 1997, Steiner 2002a). In the United States (U.S.), the Greater Boston region and the New York metropolitan area are examples of urban regions. Forman's (2008) definition of urban region includes larger areas outside the urban core, including the metropolitan area and an urban-region ring (see Figure 1.2 in Forman 2008, p. 6). An urban region is highly heterogeneous with complex multidirectional continuous and dynamic processes (e.g., land development, land abandonment, forest loss, forest regeneration, population increase or decrease, water and species movement). In summary, an urban region is a complex adaptive system (Lansing 2003, Levin 2003, Norberg and Cumming 2008). Urban regions are also where most people live in the U.S. (Hobbs et al. 2002) and the world (United Nations 2008), and often coincide with the areas of high biodiversity conservation priority (Groves et al. 2000, Balmford et al. 2001, Araújo 2003). An urban region is also a central and relevant scale of planning/design/management, especially for species such as forest birds that have a large home range and a long dispersal distance. Therefore, land-use plans for urban regions should explicitly integrate the conservation of biodiversity as a recognized priority (Ahern et al. 2006).

An urban region is a social-ecological system (Berkes et al. 2003), which is a complex adaptive system with important characteristics such as self-organization, non-linear relationships, and thresholds (Levin 1998, 1999, 2003, Lansing 2003, Norberg and Cumming 2008). Therefore, approaches and concepts such as adaptive (co)management (Holling 1978, Walters 1986, Lee 1993, 1999, Gunderson et al. 1995, Berkes et al. 2003, Olsson et al. 2004, Lister 2008), holistic thinking (Zonneveld 1990, 1994, 1995, Naveh and Lieberman 1984), complex adaptive systems theory (Lansing 2003, Levin 2003, Norberg and Cumming 2008), and resilience thinking (Holling 1973, Gunderson and

Holling 2002, Gunderson and Pritchard 2002, Berkes et al. 2003, Pickett et al. 2004, Walker and Salt 2006, Woodward 2008) have been argued to be important for planning, designing, and managing an urban region and landscapes and ecosystems within. There has been an increasing number of successful cases of adaptive management of natural resources (Walters and Holling 1990, Lee 1993, Gunderson et al. 2008, Lister 2008) and describing insect colonies, immune systems, brains, and economies by complex systems (Mitchell 2009). However, the application of the concept of ecological resilience to more human-dominated systems, especially to landscape and conservation planning, while potentially significant, is still just beginning (Alberti and Marzluff 2004, Pickett et al. 2004, Woodward 2008).

4.1.1.3 Change Is the Norm in a Social-Ecological System

Dynamic change or "surprise" in ecological and social systems is more common than many people think (Gunderson 2003, Reid 2006). Change and disturbance are not some isolated events but are often cyclic, recurrent parts of a system (e.g., a disturbance regime) (Botkin 1990, Pickett et al. 1992, Holling and Meffe 1996). Social-ecological systems change continuously—often in a constant flux. Human activities, including land use, change the landscape as well as respond to the changes made by both human and natural causes. Planning informs and influences change. Planning and design both change and respond to the spatial configuration of land use/cover and ecological patterns, and associated processes such as the flow of water, nutrients, and organisms (Forman and Godron 1986, Forman 1995, Ahern et al. 2006). If change is more of a norm than an anomaly in social-ecological systems, including urban regions, I argue, it may make more sense to plan and design landscapes and urban regions to account for unavoidable and eventual disturbances and to increase the resilience capacity of the system so that it does not flip into an undesirable state (e.g., eutrophication of a lake [Carpenter 2001], the global unrest scenario [Gallopín 2002], the “collapse” scenario [Newman et al. 2009], etc.). How can planners enhance the resilience of an urban region? How can planning and design contribute to building a capacity to work with change? I would argue that two ways to achieve this are to increase response/functional diversity (Elmqvist et al. 2003) and to create connectivity across scales (Ahern 1991, Zipperer et al. 2000, Vos et al. 2002). Before discussing these concepts, I will first review the founding concepts of resilience, adaptive cycle, and panarchy as an organizing framework.

4.1.1.4 Founding Concepts: Resilience, Adaptive Cycle, and Panarchy

4.1.1.4.1 Resilience

Resilience is defined as “the capacity (or ability) of a system to absorb disturbance and still retain its basic function and structure” (Walker and Salt 2006, p. xiii). This definition focuses on the system’s retaining the ability to recover at all after disturbance (Walker and Salt 2006). Resilience can also be defined as the capacity of a system to absorb disturbance without shifting to another regime (Holling 1973, Walker et al. 2004, Walker and Salt 2006) and this capacity is the key to sustainability (Walker and Salt 2006). Because a resilient social-ecological system (e.g., ecosystem and landscape) “has a greater capacity to avoid unwelcome surprises (regime shifts) in the face of external disturbances,” it “has a greater capacity to continue to provide us with the goods and services that support our quality of life” (Walker and Salt 2006, p. 37). Therefore, maintaining and enhancing the resilience of a system is important for the sustained provision of ecosystem services (Elmqvist et al. 2003) and ultimately, creating sustainable landscapes.

4.1.1.4.2 Adaptive Cycle

The adaptive cycle is integral to the idea of resilience, and Holling and his colleagues argue that it represents more or less a universal progression of a system (e.g., ecosystem and landscape) over time (Holling 1986, 2009). An adaptive cycle consists of four phases: rapid growth (r), conservation (K), release (Ω), and reorganization (renewal) (α) (Figure 4.1). The adaptive cycle can be understood as a metaphor for describing change in ecological, social, and social-ecological systems through time; in other words, how these systems evolve over time, and how they respond to change and disturbance (Holling 1986, Gunderson et al. 1995, Holling et al. 2002a, b). Note that the evolution of a system is not always an orderly sequence from r to α. Systems can move back from K toward r, or from r directly into Ω, or back from α to Ω (Walker et al. 2004).

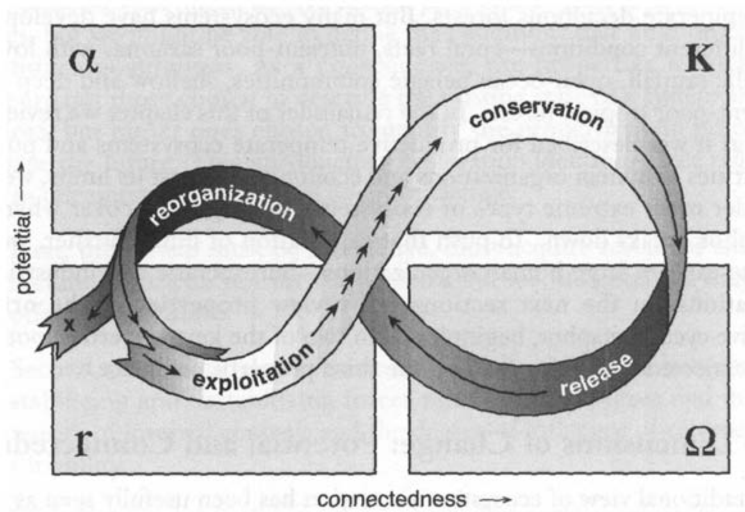

Figure 4.1: An adaptive cycle in a two-dimensional representation. Short arrows represent a slow change and long arrows represent a rapid change. Connectedness is the degree of connectedness among the variables; the more tightly connected the variables are, the more rigid a system is, and thus, the more vulnerable the system is to external disturbances. In other words, connectedness is the degree of internal control that a system can exert over external variability (Holling and Gunderson 2002). Potential is the amount of accumulated capital (e.g., nutrients and carbon) stored in dominant structuring variables at that moment in the system (Gunderson et al. 2002a). Potential sets limits to possible future options (Holling and Gunderson 2002) (*Source*: Holling and Gunderson 2002, Figure 2-1, p. 34).

In social-ecological systems such as landscapes, changes are inherent and possibly continuous. The adaptive cycle shows that a system eventually goes through the Ω phase (creative destruction). Considering the current global financial crisis, peak oil, and climate change, one may think that indeed the whole world is headed to a reorganization or “collapse” phase. It remains to be seen if we can recover at all from the collapse and restore our former economic system, or perhaps cross a threshold into another economic state. History is full of examples of the collapse of civilization due to the overexploitation of natural resources (among other causes) on which the local population relied (Diamond 2005).

The back loop is from the Ω to α phase of an adaptive cycle (Figure 4.1). It is the back loop that is critical for the maintenance of the essential structure and processes of a system, and thus, its resilience (Gunderson and Holling 2002, Walker et al. 2004). The back-loop is also important for the evolution of the system to the growth phase and the front loop (the next cycle). If a release (the Ω phase) is inevitable, important related questions include: How can the back loop be gracefully navigated? How can an ecosystem, or landscape retain, cultivate, or improve its capacity to respond to the inevitable change and disturbance while returning to a similar state that it previously occupied? These questions will be examined in the light of landscape planning.

4.1.1.4.3 Panarchy

In short, panarchy is composed of adaptive cycles linked over many scales (Walker and Salt 2006). An adaptive cycle is a representation of the evolution of one system at a certain scale. However, any system is actually composed of an integrated system of linked adaptive cycles, and they interact across space and time (Gunderson and Holling 2002, Walker and Salt 2006). This linked set of adaptive cycles is referred to as a panarchy (see Figure 4.2) and it is the interactions between the linked adaptive cycles that govern the behavior of the whole system (Walker and Salt 2006).

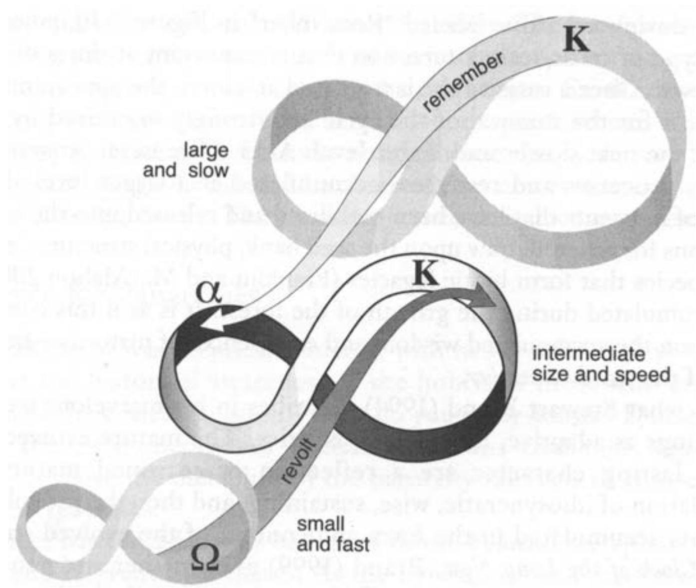

Figure 4.2: Panarchy represented by three interacting scales. "Arrows labeled "revolt" and "remember" indicate key linkages across space and time scales. Smaller-scale elements that are in the Ω phase (creative destruction) can synchronize and cascade to create a transition to the Ω phase at broader scales, as represented by the "revolt" arrow. Broader scales provide resources during smaller-scale reorganization phase, as suggested by the "remember" arrow" (Gunderson et al. 2002a, p. 15) (*Source*: Holling et al. 2002b, Figure 3-10, p. 75).

An ecological example of "revolt" is when conditions in a forest allow for a local ignition to create a small ground fire that spreads to the crown of a tree, then to a patch in the forest, and then to a whole stand of trees. A small, local fire cascades up in scales to initiate a wide-spread fire. In this example, "remember" is, after a fire in an ecosystem, processes and resources accumulated at a larger level which slow the leakage of nutrients that have been mobilized and released into the soil. The options for renewal draw upon the seed bank, physical structures, and the adaptive capacity of surviving species that form biotic legacies (Franklin and MacMahon 2000) that have accumulated during the growth of the forest (Holling et al. 2002a, b). Serotiny is the adaptation of woody plants to respond and recover from fire, for example by releasing seeds after the burn, or by resprouting foliage directly through tree bark. Even after devastating earthquakes, atomic bombs, and volcanic eruptions, the destruction is never 100 percent complete, albeit close and not to downplay the inflicted damage; there are always some “seeds” such as miraculously survived structures (e.g., the dome in Hiroshima, buildings that withstood the shocks, vegetation and trees that survived the larva), including dead structures (e.g., burnt but still standing snags) that provide seeds for recovery (Franklin and MacMahon 2000, Chen 2005). Another example of “remember” is institutional (social) memory—“the reservoir of informal strategies and the experiences accumulated by people using them” (Norberg et al. 2008, p. 66)—that can become important for dealing with infrequent disturbances and crisis (Dale et al. 1998, Berkes and Folke 2002, Folke et al. 2003, Redman and Kinzig 2003).

The processes of “revolt” and “memory (or remembrance)” are what sustain and define resilience within a system: “Resilience within a system is generated by destroying and renewing systems at smaller, faster scales. Ecological resilience is reestablished by the processes that contribute to system “memory”—those involved in regeneration and renewal that connect that system’s present to its past and to its neighbors” (Gunderson et al. 2002b, p. 258). The Ω phase (creative destruction) that follows “revolt” from a lower scale and the “remembrance,” from a higher scale, that shapes reorganization (the α phase) are the products of cross-scale interactions (Figure 4.2, Gunderson et al. 2002a).

The cross-scale dynamics of the natural and social components of a complex system are at the core of panarchy and the processes such as “revolt” and “remember” are in turn reinforced by panarchy patterns—that is, the patterns and processes are self-organizing (Levin 1999, Gunderson and Holling 2002, Walker and Salt 2006). This is a key aspect of complex adaptive systems (Levin 1998, 1999, Holling and Gunderson 2002). Panarchy is composed of nested adaptive cycles and the interactions among the different spatial and temporal scales are the key to the resilience of a system at a particular focal scale (Walker et al. 2004).

4.1.2 Propositions

Walker and Salt (2006) propose three steps to manage for and enhance resilience of a social-ecological system: step 1, to understand the drivers (i.e., slow, controlling, coarse-scale variables often coupled with fine-scale, fast variables); step 2, to know the thresholds on the drivers; and step 3, to enhance aspects of the system that enable it to maintain its resilience. To address the last step by landscape or urban planning and design, it can be broken down into two sub-steps. The first is to identify these aspects and the second is to develop a plan, scheme, or strategy to enhance the aspects by planning and design. I would argue that response/functional diversity, redundancy, and connectivity across scales are exactly the attributes of a system that are essential to build resilience capacity, which landscape and urban planning and design may help to develop, maintain, or restore. Response/functional diversity, redundancy, and connectivity across scales are the specific “handles” on which planners and designers have leverage by way of influencing land use patterns and regional development and growth.

Based on ecological theories and a resilience approach to managing complex adaptive systems, I pose two propositions and attempt to link them to landscape planning—more specifically, conservation planning for species, populations, habitats, and ecosystems—in an effort to develop a landscape planning framework for biodiversity conservation at the urban regional scale. The first proposition is that (1) response/functional diversity and redundancy and (2) connectivity across scales are key to the resilience of a social-ecological system. The second proposition is that landscape planning and design can influence these factors (aspects) of a system (e.g., a landscape and an urban region) to maintain, restore, enhance its resilience.

4.1.2.1 Response Diversity, Functional Diversity, and Redundancy

Response diversity, functional diversity, and redundancy are key interrelated, ecological concepts to be integrated in the proposed landscape planning framework for conserving biodiversity in urban regions. Little has been discussed or practiced thus far in the literature about explicitly relating these important concepts to landscape planning. By integrating these concepts into the planning framework, I argue that it can ultimately enhance the resilience of an urban region.

Response diversity has been identified to be critical to a system's resilience (Elmqvist et al. 2003, Walker and Salt 2006, Lister 2008). Response diversity in biological systems refers to the diversity of responses to external disturbances among species within the same functional group (Norberg et al. 2001, Elmqvist et al. 2003, Walker and Salt 2006). For example, there are many pollinators but if all of them respond to external disturbances in a similar way, response diversity is low. On the other hand, if there are a variety of responses, response diversity is high. Alternatively, response diversity means each species in the same "lump" (e.g., body mass) having similar scale of function but having different responses to unanticipated environmental change (Holling et al. 2002b). In general, if there are few species within functional groups, response diversity tends to be low (Walker and Salt 2006).

Response diversity is a way to compensate for lost functions within a narrow range of scales because even when one species declines or becomes extinct, if there are other species that perform the same/similar function and have different sensitivity to a particular disturbance, this ecological function is more likely to be maintained, leading to the resilience of the ecological function (Gunderson et al. 2002b, Hooper et al. 2005). Therefore, response diversity is critical for the maintenance of ecosystem processes over time, particularly during periods of reorganization after disturbance events (Daily 1997, Peterson et al. 1998, Gunderson et al. 2002b, Holling et al. 2002b, Elmqvist et al. 2003, Walker and Salt 2006). Increase in response diversity leads to decreased sensitivity to disturbances and enhanced resilience.

Functional diversity is a similar and broader concept than response diversity, and it also contributes to resilience (Gunderson et al. 2002a, b). Functional diversity in biological systems refers both to the diversity of functional groups (across scales) and to the diversity of species within functional groups (Peterson et al. 1998, Walker et al. 1999). High diversity within functional groups usually leads to high response diversity. "The within-scale and between-scale diversity produces an overlapping reinforcement of function that is remarkably robust" (Holling et al. 2002b, p. 85). Peterson et al. (1998) and Gunderson and Pritchard (2002) provided examples and empirical evidence to support the proposition that: "resilience derives from functional reinforcement across scales and from functional overlap within scales" (Gunderson et al. 2002b, p. 253). "Across-scale resilience is produced by the replication of process at different scales. The apparent redundancy of similar functions replicated at different scales adds resilience to an ecosystem (Holling et al. 1995, Folke et al. 1996, Walker and Salt 2006). Because most disturbances occur at specific scales, similar functions that operate at other scales are maintained" (Gunderson et al. 2002a, p. 10). Therefore, it can be concluded that both within- and between-scale diversity contribute to the resilience and sustainability of the system (Holling et al. 2002b, Hooper et al. 2005).

As discussed, redundancy reinforces response and functional diversity. When there is redundancy in the same functional group, response diversity tends to be high because the redundant species are more likely to respond differently to external disturbance (Walker and Salt 2006). When there are redundant functions across scales, the specific function is more likely to persist in the face of disturbances and local extinctions (Holling et al. 1995). In sum, the resilience of ecosystem function depends on both (1) the diversity of functions (and functional groups) and having these functions replicated across a range of spatial and temporal scales and (2) the diversity of species within functional groups (i.e., response diversity).

4.1.2.2 Connectivity across Scales

Hierarchy is a common organizational structure found in nature and human societies. According to the Oxford Dictionary of English, hierarchy in this research is defined as "an arrangement or classification of things according to relative importance or inclusiveness" (Soanes and Stevenson 2003, p. 817). A nested hierarchy is a particular case of hierarchy. It is a hierarchical ordering of nested sets (Lane 2006). (A set is a collection of distinct objects, considered as an object in its own right [Soanes and Stevenson 2003].) The concept of nesting is exemplified in Russian matryoshka dolls. Each doll is encompassed by another doll, all the way to the outer doll. This is the concept of nesting. When the concept is applied to sets, the resulting ordering is a nested hierarchy. In a nested hierarchy all of the ordered sets are nested. Therefore, a nested hierarchy has a more strict meaning than a (simple) hierarchy. Nested hierarchies are the organizational schemes behind taxonomies and systematic classifications (Knox 1998). The panarchy is defined as adaptive cycles that are nested one within the other across space and time scales (Gunderson and Holling 2002).

Biodiversity can be considered to exist in a nested hierarchy of the four levels of organization: genes, species-populations, ecosystems-communities, and regional landscapes (Noss 1990). A nested subset structure can be found in bird community composition if the species found in species-poor communities are also found in progressively more species-rich assemblages (Worthen 1996). Administrative/geographic units in planning and design also exist in a hierarchy from each parcel of land to neighborhood (site), city/town (community), and region (Sipes and Lindhult 2007). Hierarchy theory (Allen and Starr 1982, O'Neill et al. 1986, Allen and Hoekstra 1992, Levin 1992, King 1997) provides an important foundation on the way each scale in a hierarchy interacts with the others. Faster dynamics at finer scales give rise to slow dynamics at coarser spatial scales; the properties and behavior of individuals and species, populations, and their interactions develop patterns and processes at higher hierarchical levels (e.g., communities and ecosystems) (Wiens et al. 2002). The structure and processes of systems at coarser scales or higher levels, in turn, control the dynamics of systems that can occur at finer scales or lower levels. Each target scale, whether a unit of planning and design or a level of biological organization, interacts with the scales below and above, and functionally connected to them by feedback loops, "revolt" and "memory" (Gunderson et al. 2002a, b, Holling et al. 2002b, Figure 3-10, p. 75), the flow of water, organisms, materials, and nutrients (Forman and Godron 1986, Forman 1995, Saura and Pascual-Hortal 2007). Each level in a hierarchy and each element in the same level are connected by these processes.

Connectivity via dispersal, movement, and migration is an important process for maintaining populations over time (Haila et al. 1993, Andrén 1994, Pearson et al. 1996, Wiens et al. 1997, With and King 1997, McIntyre and Wiens 1999, Fahrig 2001). Connectivity has an emergent property that is born out of the interaction of the target ecological process (or the organism of conservation interest) with the landscape (Green 1993, Turner et al. 2001). Connectivity, more specifically, functional connectivity is determined by the movement capability of the organism and landscape structure. For example, even though both forest birds and forest small mammals inhabit forest, perceived connectivity by these organisms is different with the same physical connectivity (spatial configuration) of forest patches because forest birds are more vagile, and therefore, for them these forests are more "connected."

Connectivity can be achieved as a network as well as a hierarchical connection to other levels (Vrijlandt and Kerkstra 1994, Forman 1995). Nature is full of examples of connectivity at multiple scales such as leaf veins and a river system with the first order of streams to form the second order of streams, which in turn form the third order, etc. Note that this system exists not only in a network but also in a nested hierarchy. A network of connectivity at multiple scales enables efficient, wide-spread transport of essential nutrients and water. Human-made transportation networks (e.g., subway and ground rail lines) in a densely populated region such as the Tokyo metropolitan region mimic nature's design of connectivity in a nested hierarchy. As can be seen in these examples, the major benefit of connectivity across scales is a wide and comprehensive coverage by the network; and this is an efficient coverage because of the integration of multiple scales—a finer scale for a small area and a coarse scale for a large area.

Cross-scale connectivity relates to response diversity and mitigating/transmitting disturbances—recovering after disturbances and/or maintaining disturbance regimes. This recovering corresponds to increasing resilience: increasing the capacity to recover, or undergo some change but still retain essential structure and feedbacks (Walker et al. 2004, Walker and Salt 2006). For example, I propose that redundancy of landscape elements at each scale and across scales is a way to increase resilience with the trade-offs of increased maintenance cost and cost to restore/develop these elements. An important caveat here is that connectivity can also spread undesirable disturbances such as disease, pest outbreaks, and fire (Simberloff and Cox 1987, Rosenberg et al. 1997, Bennett 1999). The amount and spatial configuration of the relevant elements (e.g., fuels) influence the way disturbances spread (Turner et al. 1989, Turner and Romme 1994). Therefore, simply increasing the connectivity of open spaces, for example, sometimes helps these unfavorable disturbances to spread. Due caution needs to be exercised when deciding where to protect, restore, and create open space, and the decision needs to be based on the evaluation of its relative importance to the target ecological process and against costs (Rosenberg et al. 1997, Beier and Noss 1998, Bennett 1999).

Considering the concept of connectivity for target ecological processes helps landscape planners decide, for example, how to best place or manage greenspaces in urban environments (Flores et al. 1998, Zipperer et al. 2000). Different levels of connectivity provide a useful conceptual framework to organize greenspaces across scales and challenge landscape planners to achieve connectivity in this manner. To address the hierarchical nature of biological organization and administrative/geographic units of planning and design, the landscape planning framework aimed for conserving biodiversity in urban regions needs to necessarily take a multi-scale approach. The biodiversity conservation planning framework needs to address multiple scales—mostly spatially at a neighborhood-, city-, and regional-scale, but temporal scales are also important.

4.1.3 Summary

I have argued that response/functional diversity, redundancy, and connectivity across scales are indeed the aspects of a social-ecological system that enable it to maintain its resilience. I argue that they have particular relevance to biodiversity conservation planning. In the next section, I will discuss how planning and design can integrate these important ecological concepts into a landscape planning framework for biodiversity conservation at the urban regional scale.

4.2 Model Development

Increasing the resilience of social-ecological systems, including urban regions, ensures the likelihood that valuable ecological (ecosystem) processes will be maintained into the future. Given inevitable change and disturbance, how can planning and design enhance the resilience of urban regions? How can planning and design cultivate or improve the capacity of an urban region to provide ecosystem services over time in the context of change? To do so requires an understanding and analysis of an urban region for which a landscape planning model is developed. Carpenter et al. (2001) and Walker and Salt (2006) recommend analyzing the system of interest for the resilience "of what, to what": for example, "the resilience of the Everglades vegetation to fires and droughts (as phosphate levels increase)" (Walker and Salt 2006, p. 120). In this example, fires and droughts are the drivers of the system. The drivers of the system are key slow variables with thresholds (Gunderson and Holling 2002, Gunderson and Pritchard 2002, Walker and Salt 2006). Walker and Salt (2006) has further proposed a systematic method to understand and analyze the social-ecological systems, and identify the aspects of a system to manage it for enhanced resilience. They propose that: first, understand the slow, controlling drivers; second, know the thresholds on the drivers; and third, enhance aspects of the system that enable it to maintain its resilience. This method is based on years of study on various ecological systems and natural resource management—and has recently been applied to social systems as well—by Holling and his colleges at the Resilience Alliance (Holling 1973, 1996, Gunderson and Holling 2002, Gunderson and Pritchard 2002, Berkes et al. 2003, Folke et al. 2004). I argue that this general framework can be adopted by landscape and urban planning to develop a landscape planning method to enhance the aspects of an urban region that enable it to maintain its resilience.

In the context of conserving forest birds in an urban region, I argue that the percentage of forest in an urban region is a key slow variable with area-based threshold(s), which affects the resilience of forest bird populations. Extinction, colonization, parasitism, nest predation, vegetative complexity, life histories are among other factors that affect forest bird abundance, diversity, and species composition (Marzluff 2001, 2005, Miller et al. 2001). Even if an explicit threshold amount cannot be detected, many studies have found forest area or the percentage of forest cover to be a critical variable affecting forest bird species richness, abundance, and occurrence (Mörtberg 2001, Alberti and Marzluff 2004, Huste and Boulinier 2007, Moning and Mueller 2008, Caprio et al. 2009). When a threshold is detected, the rate of decline in bird abundance changes differentially above and below the threshold forest amount or percentage. At the urban regional scale and among the key variables controlling the dynamics of forest bird populations, the percentage of forest, governed by background tree growth rate, is the slowest variable. It is on the scale of many decades for establishment, growth, maturity, and decline, but it is coupled with faster variables such as land use change and land conversion, which occur on the time scale of a few years. For example, forest clearing, suburban sprawl, and conversion of forest to agricultural land and housing are factors affecting the percentage of forest as well as spatial configuration of forest in a landscape (Figure 4.3). Other factors that affect the amount and spatial configuration of forests include conservation and restoration efforts, and natural disturbances such as wind and beavers. Moreover, climate change, operating at the global scale, affects urban regional forests. These factors have variable rates of change. For example, forest recovery (natural regeneration and tree planting) after agricultural abandonment in New England in the U.S. took 100-150 years (Foster et al. 1998), whereas conversion of forest to housing may take only a few years. In sum, the slowest variable is the percentage of forest in an urban region, and finding a threshold, if it exists, means how much disturbance, such as forest loss and fragmentation due to land conversion, the system (i.e., the urban region) can take before it flips to another regime, where forest bird abundance is drastically low. The alternative regime would have negative effects on the functioning of natural systems including forest ecosystems with lower production of forest goods and services that are tied to human-well being. For example, lowered connectivity of natural cover and amount, worsened air pollution, lower water quality, increased heat island effect, and lowered biodiversity, may be expected (Vitousek 1997, Alberti and Marzfull 2004, Folke et al. 2004, Grimm et al. 2008).

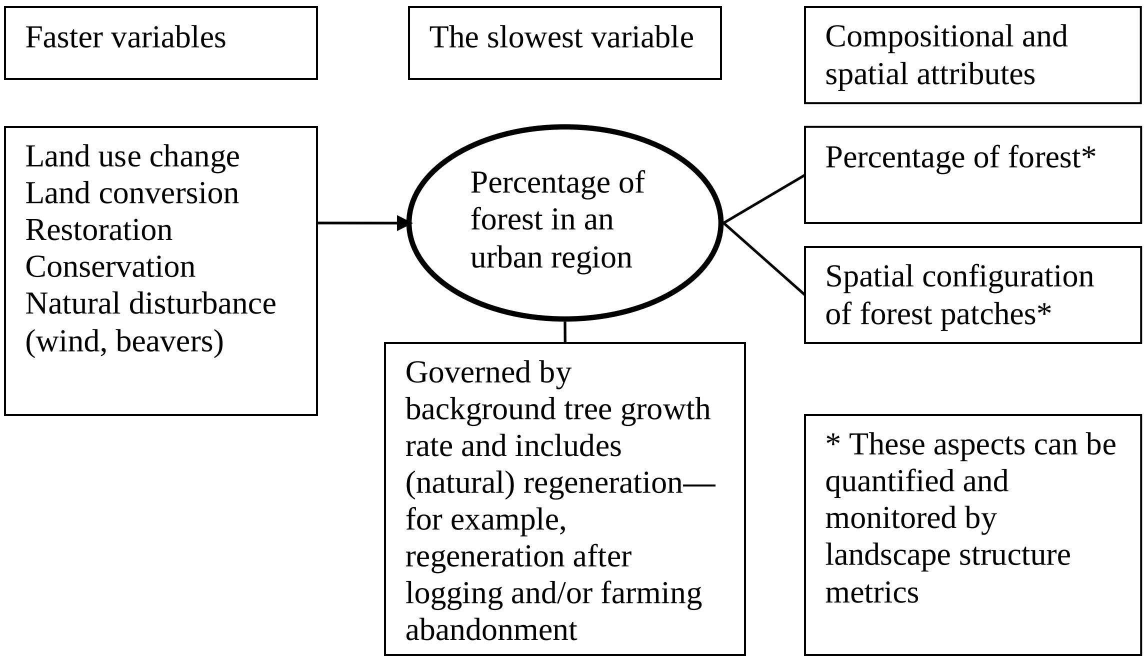

Figure 4.3: Schematics of the variables that affect the slowest variable of interest at the urban regional scale. Note that the background tree growth rate encompasses much faster variables in different levels of biological organization, such as needles and leaves, and phenological changes of trees.

4.2.1 How Can the Concepts of Response and Functional Diversity, Redundancy, and Connectivity across Scales Be Translated to Landscape Planning and Design?

I argue that planning and design can contribute to increasing the response diversity of a landscape for specific functions. In landscape planning, response diversity translates to having many landscape components that perform the same function. For example, if the function of concern, or the goal of planning, is species conservation, stepping-stone habitats, corridors, large patches, and small patches are all landscape components that can provide habitat for some target species (Table 4.1). Each landscape element, however, would respond differently to change and disturbance. Therefore, having a variety of landscape elements is important to increase response diversity even though they may perform the same function.

In the context of conserving forest bird species and populations in an urban region, although any landscape element with a certain amount of forest cover (Table 4.1) may be able to contribute to maintaining bird populations, green landscape elements differ in important ways: some have interior habitat and a complex spatial structure, for example. Even though there may be redundancy in an urban region's green spaces (open spaces with tree covered areas) in terms of their ecological functions (e.g., air and water purification, pollutant removal, water retention, and microclimate amelioration), each structure or landscape element such as bioswales, permeable parking, rooftop gardens, and protected forests would respond differently to change and external disturbance—thereby increasing response diversity. Therefore, when planning an urban region for forest bird conservation, not having many of the same kind of conservation measures such as only protected forests, but a variety of different "forest types," such as protected forests on a public land, a restored grove, street trees, and trees in parks and private gardens is important for the resilience of an urban region for the conservation of forest bird species and populations. Different ownership types, vegetation composition and vertical structural characteristics are significant for establishing new green spaces and managing them at the neighborhood and city scales.

To increase response diversity at the urban regional scale, tree cover should be in different size classes, vertical structure, species compositions, and spatial configurations. Trees have many functions (e.g., air and water purification, stormwater runoff reduction, wildlife habitat, aesthetics, provision of shade, microclimate mediation, etc.) (Spirn 1984, Nowak et al. 2006, Jim and Chen 2008, American Forests 2009). Although the effectiveness in providing these functions differs depending on specific tree species, and vegetation composition and structure, different amounts and spatial configurations of trees—such as street trees, remnant forests, and trees in parks, on university campuses, in orchards, in cemeteries, and in riparian areas—can provide these functions (Table 4.1). A related strategy of increasing the response diversity of regional tree canopy cover may be to maintain forests in different life cycle stages, for example, vigorously growing, maturing, and declining—relating to the different phases in the adaptive cycle (White and Pickett 1985, Noss and Harris 1986). These forests in different growth stages can provide different habitats for different forest species and spread the risk of disturbance such as insect outbreaks and storms across a landscape. It can be concluded that increasing response diversity, in this species conservation context, means to have a variety of different habitat types (e.g., a variety of ecosystems), different growth stages of each habitat type, and different composition and spatial configuration of landscape elements in a landscape.

The significance of Table 4.1 is that different levels of biological organization are matched with relevant planning scales for management and planning. For example, metapopulations can be best managed and planned for at the urban regional scale; local populations can be best targeted at the city scale. Genes are affected by the migration of individuals between subpopulations at the neighborhood scale. When migration occurs on a broad landscape, physical barriers such as highways and mountain ranges hinder the movement of organisms. Therefore, gene flow is often managed at the urban regional scale for wide-ranging organisms such as forest birds.

Colding's (2007) ecological land-use complementation (ELC) is a concept based on increasing complementation and supplementation by clustering different types of open spaces to provide emergent functions, which would not be supported if specific, individual open spaces were separated in a heavily urbanized matrix. Colding (2007) argues that clustering different types of green areas (e.g., orchard, remnant forest patches, cropland, golf courses, etc.) in the urban environment can create a synergy to support biodiversity and ecosystem processes, such as seed dispersal and pollination, by the increase in overall connected habitat and through the mechanisms of landscape complementation and supplementation (Dunning et al. 1992). If different types of urban open spaces are isolated, Colding (2007) argues, they would not be able to support ecological processes essential for biodiversity. For example, when a golf course and a forest patch are adjacent to each other, an amphibian species that needs both a pond to breed and an upland to spend the adulthood may be able to complete its lifecycle. The likelihood of survival of this species would be much lower if these two land uses were isolated in a heavily developed urban matrix. In sum, it can be said that through thoughtful planning of the spatial configuration of different types of urban open spaces, ELC can support emergent ecological functions and increase biodiversity.

In landscape planning for conservation of biodiversity, the concept of functional diversity can translate to, for example, replicating connectivity across and within scales (e.g., Li et al. 2005a). Creating a linked system of connectivity across scales is a good practice to increase resilience just as a linked network of linear open spaces such as greenways are a good example of a connected network for recreational or hydrological functions (Fábos 2004). For example, the concept of cross-scale connectivity explicitly applied to Jim and Chen's (2003) Nanjing plan. The Nanjing plan (Figure 10 in Jim and Chen 2003) shows how different open spaces at each scale are spatially connected to different open space elements at other scales. For example, the streets lined up with street trees would form a green corridor, which in turn is part of and are connected to green wedges. Green infrastructure is a collection of cross-scale replication of green practices, for example, from rain barrels, bioswales (street level) to neighborhood parks, and connected networks of parks, open spaces (e.g., cemeteries, orchards, agricultural lands, remnant forests) at a regional scale. In another example, Opdam et al. (1993, 2003) applied the concept of metapopulations to conservation planning and developed the concept of landscape cohesion. They then used the concept of landscape cohesion to develop ecological networks (Opdam et al. 2006). Their ecological network concept allows its constituent elements (e.g., patches, stepping-stone habitats, corridors) to change (e.g., developed and/or spatially change the locations) while maintaining the goal of providing a conservation network for species (Opdam et al. 2006). This is an example of providing a durable framework for species conservation: having structures (i.e., landscape elements supporting conservation goals) located across a landscape at multiple scales would allow ecological processes and biodiversity to persist despite a variety of disturbances (e.g., habitat loss, land conversion, natural disasters), each occurring at different scales, due to response and functional diversity provided by the constituent elements.

In planning, a group of landscape elements belonging to a certain scale is akin to species with similar body mass that occupies a particular niche at a specific scale. For example, green spaces at the regional scale may exist as a connected system of regional parks, riparian corridors, stepping-stone habitats, large protected forest patches, etc. The function of providing habitat to the selected forest birds and facilitating dispersal and movement can be replicated across scales (e.g., neighborhood, city, and region) by different groups of green space elements belonging to each scale (see Table 4.1). This is how cross-scale functional redundancy is achieved. Then, within each scale, a diversity of function can be provided by a group of landscape elements belonging to the specific scale. For example, the functions provided by regional parks include recreation, amenity, and environmental education along with habitat value to forest birds (Figure 4.4). Peterson et al. (1998) argue that resilience derives from both a diversity of function at specific scales and the replication of function across a diversity of scales.

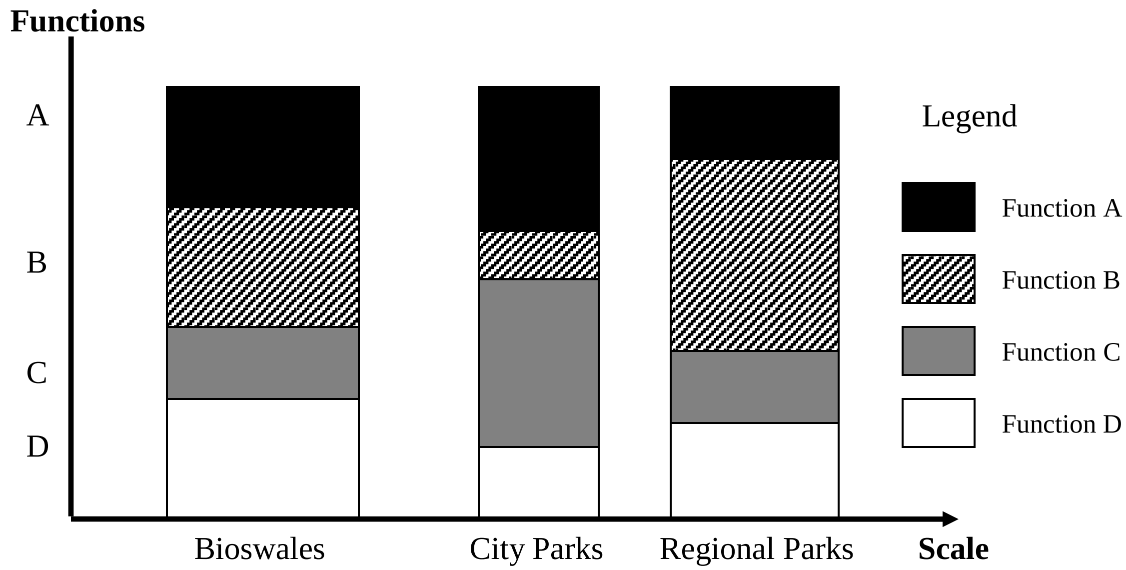

Figure 4.4: Conceptual model representing the relationship between the scale of planning and various functions that landscape elements belonging to each scale can provide. Bioswales, city parks, and regional parks are examples of green space at the neighborhood, city, and regional scale, respectively (see Table 4.1 for other green landscape elements). These green elements can provide a diversity of functions such as facilitating animal dispersal and providing habitat for forest birds, recreation, aesthetics, and environmental education. Different patterns of shading represent different functions provided. Note that the proportion of functions within and across scales is not scaled to the actual percentage of each function provided. Within scales, the presence of different functions provides robust functioning, whereas replication of function across scales reinforces function. Both mechanisms maintain the resilience of a landscape (modified from Figure 9 in Peterson et al. 1998 and Figure 1.2 in Gunderson et al. 2002a, p. 11).

Multifunctionality, therefore, becomes the key to increase functional diversity at specific scales (and therefore response diversity) when functions other than ecological (ecosystem) are considered. Because the number of functions provided (a measure of multifunctionality) differs depending on temporal and spatial scales, it is important to consider multifunctionality at each scale (Priemus 2001, Rodenburg and Nijkamp 2004, Louw and Bruinsma 2006). At the urban regional scale, for example, ecological functions as well as transportation, housing, culture, and economic development are important (Forman 1995, 2008). At the scale of an agricultural field, planting a variety of crops can increase crop resilience to diseases and insect outbreaks, and preserving hedgerows would facilitate small mammal’s movement, provide shelter for some birds, and protect soil erosion (Forman and Godron 1986, Forman 1995, Jobin et al. 2001, Altieri 2004). As part of a larger landscape, the hedgerows may act as an icon for cultural landscape identity (Vos and Meekes 1999). Identifying and planning multifunctional green networks at every planning scale (Girling and Kellett 2005) would therefore increase the resilience of a whole landscape.

Generally speaking, these planning concepts and strategies are a way to deal with inevitable surprises and disturbances and to increase the resilience of a landscape. Variability in the landscape is the key to maintaining renewal capacity when the landscape undergoes some change (Holling et al. 2002b). Planning and design should develop landscapes that are (1) more spatially heterogeneous—not fragmented but integrated at multiple scales—by adding fine-scaled elements such as hedgerows, wind breaks, pockets of nature restoration, diverse crops, integration and preservation of cultural/historical heritage (stone walls, monuments), etc. and (2) more functionally diverse (i.e., multifunctional landscapes). By doing so, response and functional diversity will be increased, making a landscape less sensitive to disturbances and building its capacity for enhanced resilience (Holling and Gunderson 2002).

4.2.2 Landscape Planning “Best Practice” Model

Although the landscape planning model I propose focuses on biodiversity conservation at the urban regional scale, several recent general landscape planning models (i.e., Steinitz 1990, Steiner 1991, Ahern 1999, Leitão et al. 2006, Kato and Ahern 2008) were reviewed first to identify important steps of planning in general, and important features and concepts to be included in the model. The reviewed landscape planning methods are comprehensive, addressing abiotic, biotic, and cultural features of a landscape, trying to be all-inclusive but necessarily general, and arguably applicable to multiple scales and in various geographical settings. They share several key features: (1) being iterative and cyclic, (2) having adaptive components (monitoring and continuous evaluation, integration of new knowledge generated through the process), (3) being transdisciplinary (Tress et al. 2005), involving the public and stakeholders throughout the planning process, (4) being applicable to multiple scales, although most applicable to the landscape level (Leitão and Ahern 2002), and (5) being applicable across a range of strategic planning and abiotic-biotic-cultural contexts, as they should be because of their generalized nature.

Based on the review of existing landscape planning models, key ideas and concepts were extracted and consolidated into the following landscape planning "best practice" model (Figure 4.5). The model builds on Ahern's (1999) framework model and its influence on my thinking is acknowledged here. The best practice model has the following general steps: (1) goal setting, (2) alternative future scenarios, (3) plan development, (4) plan implementation, (5) monitoring, and (6) evaluation. The planning process is cyclic, iterative, and interactive—common in the reviewed models.

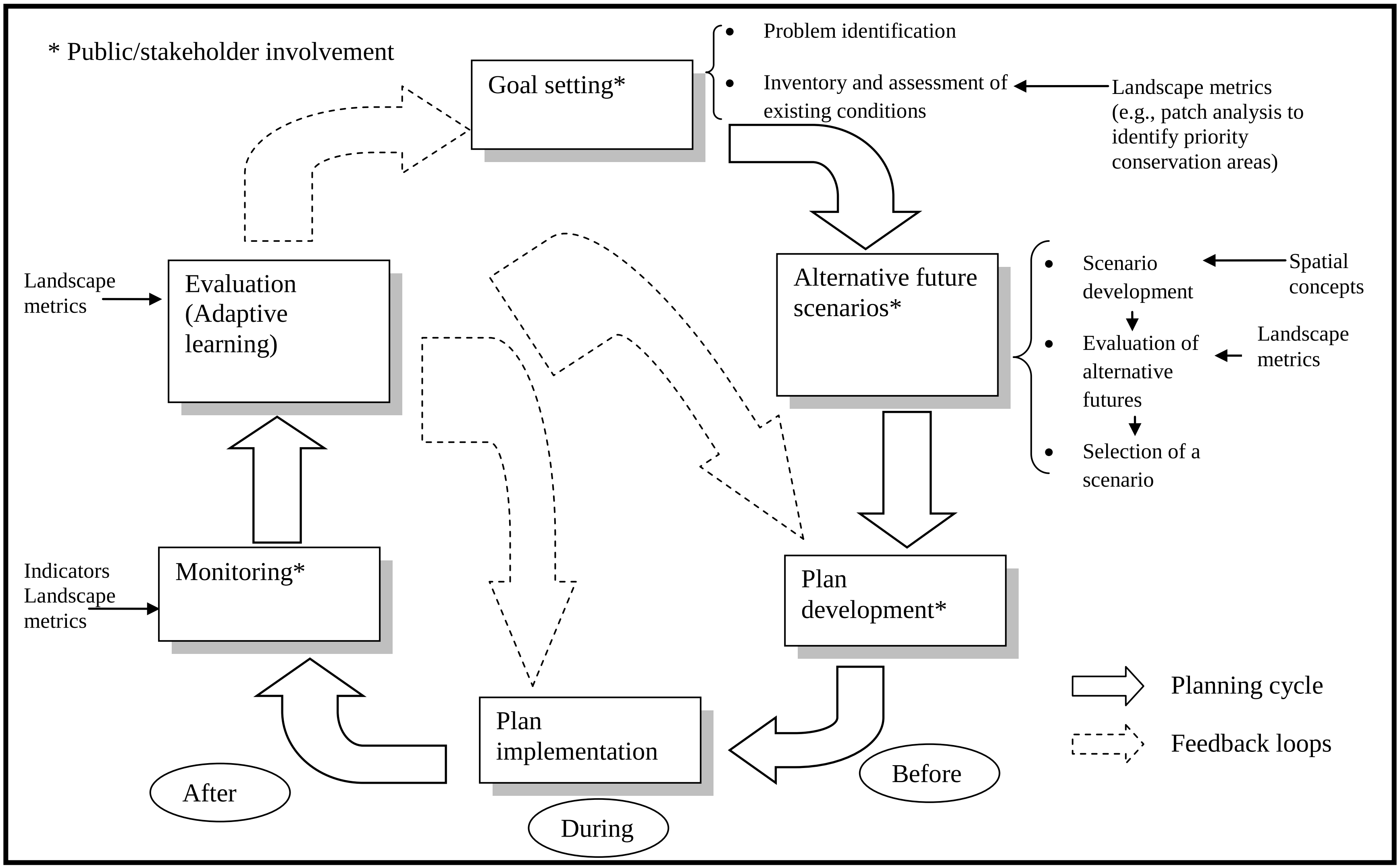

Figure 4.5: Landscape planning "best practice" model. The outside frame refers to a social-ecological system such as an urban region in which planning occurs. Dashed arrows denote feedback loops. Monitoring is an integral part of adaptive learning.

In goal setting (step 1), problems are identified and existing conditions are assessed. Multiple planning goals can be pursued at the same time. Public input and spatial concepts are used to develop scenarios (step 2). Alternative futures are developed according to the scenarios. Scenarios are "plausible stories of what might unfold in the future" (Mulvahill 2003). They can be used to explore "what-if" questions. Scenarios should include a description of the present situation, a number of alternative futures, and the necessary steps or actions needed to link the present with the future (Nassauer and Corry 2004, Steinitz et al. 2003). Each scenario corresponds to a specific policy objective—for example, a build-out scenario (maximum development), maximum conservation of natural areas/open space, and somewhere in between (Steinitz et al. 2003). The implications of proposed policy become apparent through scenarios. Then, the consequences of each scenario are evaluated against the plan goals (e.g., biodiversity, water quality, accommodating population growth) and the result of the evaluation is used to decide which alternative future is most desirable, thereby informing a planning process. Landscape metrics can be used to evaluate the consequences of alternative futures on landscape composition and spatial configuration, that are, in turn, related to abiotic, biotic, and cultural resources and processes, and with public/stakeholder input, a scenario is selected. Because in practice, scenarios are often used to explore extreme cases (what-if questions), the selected scenario may actually be the integration of both innovative and feasible aspects of all the scenarios explored. Then, an "adaptive plan" is developed (step 3) based on the selected scenario.

Monitoring (step 5) should be conducted before, during, and after the plan's implementation (step 4). Landscape metrics can be used here, too, as indicators to be monitored. Landscape metrics can be used to monitor land use change and as proxies for landscape values such as biodiversity, cultural heritage, and people's landscape experience (Dramstad et al. 2001, Leitão and Ahern 2002). Monitoring is an integral part of adaptive planning (Kato and Ahern 2008) where the result of monitoring is used to inform the planning process. Based on the results, the plan and associated policy are adapted to achieve the intended result, and even plan goals may be changed (feedback loops denoted by dashed arrows in Figure 4.5).

Under an adaptive approach to planning, various uncertainties (e.g., determining appropriate systems or populations of study, spatial-temporal scales, and geographic extent) can become part of adaptive hypotheses (Kato and Ahern 2008). Planning and management decisions can be re-conceived as experiments and can be implemented as adaptive plans (Ahern 2004). Planners can minimize uncertainty through a monitoring program which is itself adaptive in nature, allowing them to understand the consequence of planning actions over time.

Adaptive learning (step 6) is encouraged by adapting the plan, its implementation, and even the original goals based on the evaluation of monitoring results. The best practice model facilitates the integration of the lessons learned into the existing planning goals/objectives and therefore plans themselves (the concept of "learning by doing"). By so doing, it can facilitate the continuous generation of new knowledge in a truly transdisciplinary mode, addressing abiotic, biotic, and cultural resources in a holistic, integrated way. An adaptive approach to planning is key to the best practice model.

4.2.3 Development of a Meta-model

4.2.3.1 Scope and Application Scale of the Meta-model

The concept of connectivity across scales informs a model that is applicable to multiple scales. At each scale, the model addresses the amount and spatial distribution of green landscape elements, and attempts to build functional connections among them for specific landscape planning purposes, for example, clean water and air, wildlife habitat, recreation, and aesthetics. Green landscape elements that consist of open spaces and vegetation (trees and shrubs) include, for example, hedgerows, remnant forests, orchards, cemeteries, riparian vegetation, vegetated swales, street trees, and trees/shrubs in city parks (see Table 4.1). At the same time, the model connects target scale dynamics to the scales above and below—cross-scale interactions.

Although the model was developed for application to urban regions, the important concepts used in the model and the features of the model can be applied to other scales such as the neighborhood and community scale. The model informs more optimal spatial configurations (Forman 1995, 2008, Collinge 2009), via use of landscape metrics and spatial criteria such as the minimum nearest neighbor distance, of forest patches in urban regions to support greater biodiversity and ecosystem services. Relevant issues are different at the urban regional scale as opposed to a site or neighborhood scale, for different issues become important at different scales (Wiens 1989). In terms of the spatial configuration of green space, at a neighborhood scale, vegetation species composition and vertical structure may be more important; at an city scale, clustering of different types of open spaces such as golf course, orchard, and home garden may be more important (see Colding 2007).

The model takes a multi-scale approach to address the hierarchy of biodiversity. The multi-scale approach is needed to achieve biodiversity conservation in urban regions because:

(1) Biodiversity encompasses multiple levels of biological organization from genes, populations/species, communities/ecosystems, to regional landscapes (Noss 1990);

(2) In planning and design, it is necessary to consider at least three scales to understand the larger context and details (mechanisms) affecting the plan and project of interest at the target scale (Allen and Starr 1982, O'Neill et al. 1986, Dines and Brown 2001); and

(3) Most disturbances operate at specific scales, to maintain and enhance resilience of a system, planning strategies such as protecting landscape elements that support similar functions across scales would maintain the function even when the elements at one scale are modified or destroyed by the disturbance (Gunderson et al. 2002a).

With regard to the last point, Opdam et al.'s (2006) ecological network concept, accounts for metapopulation dynamics and spreads the local extinction risk across the network by allowing the spatial configuration of network composing elements to change while maintaining the goal of conservation of multiple target species. This is an example of addressing change at the same landscape scale within the network. While the ecological network focuses on the landscape elements (e.g., stepping-stone habitats, a river corridor, remnant patches, etc.) composing the network at the landscape scale, the Casco (or Framework) concept plans the entire landscape, considering the rate of change of each land use and physical structure within a durable or stable framework (Kerkstra and Vrijilandt 1990, van Buuren and Kerkstra 1993, van Lier 1998). Casco builds a framework on slow changing important resources such as groundwater reservoirs (van Buuren and Kerkstra 1993). Within the “slow” spatial framework, more dynamic land uses are accommodated to respond to the time frame of market forces and trends (van Buuren and Kerkstra 1993, Ahern 1999). Since planning and design in the U.S. more often occur at finer scales (e.g., the neighborhood scale and the city scale) than at the urban regional scale, a landscape planning model aimed at conserving biodiversity at the urban regional scale needs to address these other scales as well.

4.2.3.2 Model Description

Traditionally, planners have addressed plan development (e.g., master plan, conservation plan, subdivision development plan, and water resource plan) at predominantly one scale (i.e., project scale). Gradually, they have recognized the importance of the plan’s larger context (the scale above) and the details (the scale below) or interactions that give rise to the pattern of the focal scale. However, simply giving it thought to the scales above and below is not enough to truly integrate the interactions across scales affecting the focal scale. Planners need a new conceptual framework to work with, which would show them how each scale is related to one another in a nested hierarchy and what tools or concepts can be used to link one scale to another. To this end, I present a meta-model (Figure 4.6). It can be used to apply the concept of cross-scale connectivity to regional conservation planning.

Indeed, a weakness and what has been lacking in previous landscape planning models is this explicit reference to other scales and how exactly to relate to these other scales—the methods, tools, and concepts that can be used to relate planning goals and recommendations to other scales. Planners acknowledge the larger context of a specific project or plan and the details of the plan in the analysis and assessment stage of a planning process; planners also consider the implications of the developed plan to these other scales and may even assess the effects after plan implementation. However, this is often the extent of consideration of other scales. Earlier planning models lacked explicit mechanism or method to integrate other scales. In short, other scales were simply acknowledged but not addressed beyond the acknowledgement. There was never a model that explicitly integrated other scales. The model proposed (Figure 4.6), at least conceptually, is the first of its kind in this respect.

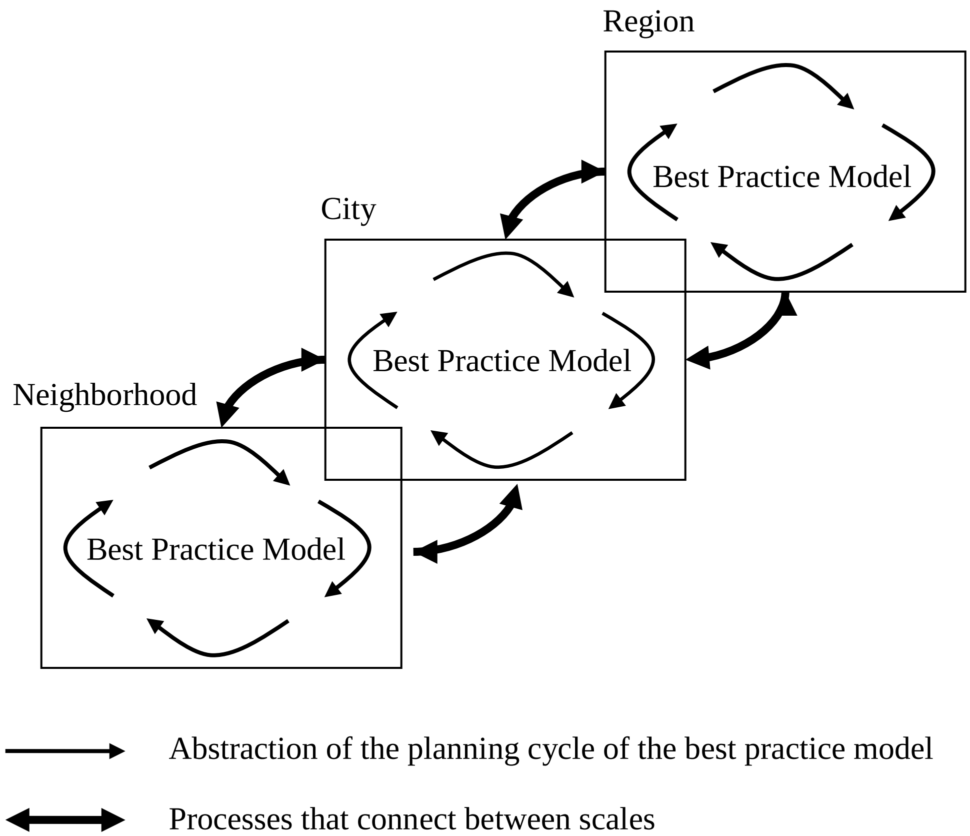

Figure 4.6: Meta-model showing the relationships and interactions among nested scales of planning unit. Thin arrows within each scale represent a planning cycle and thick arrows represent various interactions between scales. A planning process such as the best practice model (Figure 4.5) occurs at each scale. Examples of the interactions and planning concepts that can be used to link each planning process across scales include "revolt" (Gunderson and Holling 2002), institutional memory (Norberg et al. 2008), the mobile link concept (Lundberg et al. 2008), and the ecoprofile approach (Opdam et al. 2008).

Figure 4.6 is a meta-model, a model of models, which helps planners recognize cross-scale processes that are operating, organize their thinking on these processes, and identify the processes that are particularly important to their plan or project at hand. At each scale, a planning process such as the best practice model (Figure 4.5) can and, I argue, should occur. Similar to stormwater best management practices, the best practice model is a synthesis of practices that have been argued to achieve a more collaborative, adaptive, transdisciplinary, and environmentally-friendly planning (Steiner 2000, Leitão and Ahern 2002, Kato and Ahern 2008, Lister 2008). For example, Street Edge Alternatives (SEA Streets) Project in Seattle, Washington, is a successful street-scale, design experiment (Lister 2008) to treat stormwater on site as much as possible, also achieving multiple functions such as traffic claming, aesthetics, and public awareness (Water Environment Research Foundation 2008, Seattle Public Utilities 2009). The focus of the meta-model is on the processes that link different scales, represented by thick arrows (Figure 4.6). Since ecological processes such as seed dispersal and pollination transcend administrative boundaries, they are good examples of these processes that connect different scales. Other examples include "revolt" (Gunderson and Holling 2002), institutional memory (Norberg et al. 2008), the mobile link concept (Lundberg et al. 2008), and the ecoprofile approach (Opdam et al. 2008).

The mobile link concept focuses on spatial processes that link resource patches that are physically disconnected and resource patches of different types (Lundberg and Moberg 2003, Lundberg et al. 2008). It is a framework for choosing focal species (i.e., the mobile link species [Lundberg and Moberg 2003] that actively move in the landscape and connect habitats in space and time) and can be applied to the management of ecosystems that are important for producing ecosystem services in fragmented landscapes. The argument is that certain species are considered to hold the key to important ecological processes such as seed dispersal and pollination and by protecting these species these processes will be protected (Lundberg et al. 2008).

In the ecoprofile approach, for each ecosystem type such as forest and grassland, the ecoprofile of certain species is developed, using species dispersal capacity and species area requirements (Opdam et al. 2008). The ecoprofile approach can be used to plan an ecosystem network for each ecosystem type targeted for a suite of species (Opdam et al. 2008). Here, dispersal is the process that links different scales.

At each planning scale in the meta-model, Walker and Salt's (2006) three steps to manage for resilience is applicable. However, "slow," "coarse-scale" variables are relative to the scale of primary interest, for in general, the larger the spatial scale is, the slower the variables operate (Pickett et al. 1992, Gunderson and Holling 2002). Also, key slow, controlling variables differ depending on the goals of landscape planning. Once important slow, coarse-scale variables are identified at the primary scale of interest, planners need to investigate whether or not thresholds exist on the variables, and if they do, where the thresholds lie. Then as the third step, planners need to identify the aspects of a plan/project that enable it to maintain the resilience of the system in which the plan exists, and planners need to think about how planning and design can enhance these aspects.

The panarchy concept (Gunderson and Holling 2002) has shown that the interactions across scales are actually key to the maintenance and management of the resilience of a social-ecological system. In conservation planning applications, the focus of the meta-model is on processes that "connect" even physically separated patches and patches of different types, not to mention physically connected patches both across and within scales. Examples of ecological processes include the flow of water, organisms (dispersal and migration), and nutrients (nutrient cycling). Based on the panarchy concept, "revolt" and "remember" are key linkages across space and time scales (Gunderson et al. 2002a, Holling et al. 2002b). Revolt is a phenomenon whereby finer-scale landscape elements that are in the Ω phase (creative destruction) can synchronize and cascade to create a transition to the Ω phase at coarser scales (Gunderson et al. 2002a). Remember is a phenomenon whereby coarser-scale elements provide resources such as seed banks during a finer-scale reorganization (α) phase (Gunderson et al. 2002a). In biological systems, remember draws on biotic legacies (Franklin and MacMahon 2000) that have accumulated during the growth phase, connecting a system's past to its present in regeneration and renewal after disturbance (Gunderson et al. 2002b, Holling et al. 2002a). In social systems, remember can take a form of collective memory and informal strategies shared among the members of a society to deal with crises such as disease epidemics and major natural disasters (Dale et al. 1998, Berkes and Folke 2002, Folke et al. 2003, Redman and Kinzig 2003, Norberg et al. 2008).

An example of how the issues, assessments, and plans are coordinated and integrated across scales may be the use of a regional planning vision to coordinate planning efforts at the lower jurisdictional levels. A regional government or consortium may develop a regional growth and conservation plan which can guide the planning in composing municipalities and/or counties. The regional plan or vision can be used to achieve a specific goal such as developing regional networks of trails and greenways (Metro 2003). The regional plan can guide green network planning at the city, neighborhood, and site scale. The regional-scale plan provides a framework (big picture), within which each city, watershed, and neighborhood can create a vision for a greener future (Girling and Kellett 2005). Then, cities and neighborhoods are encouraged to work together with the regional government and to have their green spaces connected to the regional framework, using key ecological processes and linkages such as the flow of water and species movement, public rights-of-way, and bikeways.

In sum, there are three key concepts to Figure 4.6: (1) nested planning process showing a panarchical (Gunderson and Holling 2002) relationship; (2) different forces (external disturbances) are more influential at different scales; and (3) the importance of processes that connect across scales. The focus of the meta-model is on the interactions of hierarchically linked planning units and planning processes across space and time through the processes that link different scales. In conservation planning, the conservation of these processes rather than "objects" (e.g., species, specific sites) should be the focus of natural resource management and planning activities (Pickett et al. 1992). The meta-model is a tool for planners to organize their thinking on the cross-scale relationships and to think more clearly about them.

4.3 Model Application

4.3.1 General Application of the Meta-model

Assuming that a prior plan is in place which has guided the establishment of spatial and institutional features to build resilience capacity, an example of the general application of the meta-model is given on disaster recovery planning. Let us suppose that a major earthquake has struck a city center. At the city scale (the focal scale), resources that help recovery include miraculously standing structures and survivors—"seeds" of hope. Their spatial distribution may be spatially heterogeneous. Social capital and NGOs would contribute to the recovery. The processes, represented by the thick arrows in Figure 4.6, that help recovery include: (1) from the coarser scale, international and national aids and organized rescue efforts and (2) from the finer scale, mutual help, self-help organizations, neighborhood associations, and social movements (Davis 2005). In ecological planning, these are analogous to accumulated biological legacies that provide resources during renewal and reorganization, ecological concepts such as the mobile link concept (Lundberg et al. 2008) and the ecoprofile approach (Opdam et al. 2008) that focus on ecological processes that transcend across scales, and fine-scale phenomena that spread to coarser-scales to make a large impact.

The meta-model can also be applied to explain the interactions among different planning scales in the context of top-down planning, which is not common in the U.S. but is more common in other countries such as Japan and China where the national and regional governments have a stronger authority and power to plan. As a general application example, let us take an example of establishing an overall tree canopy goal for a region (the focal scale, this time). First, working through the regional council of governments, an overall tree canopy goal for the region can be established. This goal can be a part of a regional green infrastructure plan. Second, local governments can use the regional goal as a framework to set their own local canopy goals (Kollin and Schwab 2009). Third, local canopy goals can be further stratified by land use. If canopy is lower in a certain land use, higher goals can be set for other land uses that can accommodate more tree cover to reach the overall citywide or regional canopy goal (Kollin and Schwab 2009).

4.3.2 Conservation Planning Application of the Meta-model

I have argued that in the context of conservation planning, providing response/functional diversity and connectivity across scales would increase the resilience of an urban region (see section 4.2.1). In conservation planning practice, these two concepts can be merged into a single goal of creating (functional) connectivity of landscape elements that can provide response/functional diversity across scales. Since the method for increasing response/functional diversity has been described (section 4.2.1), here I will focus on the method for providing connectivity across scales using green spaces with different amounts and configurations of tree cover (see Table 4.1 for what landscape elements would constitute these green spaces) for the purpose of maintaining regional forest bird populations. Connectivity facilitates the key function of my focus, dispersal. Dispersal is a key mechanism for maintaining metapopulations (Levins 1970), source/sink dynamics (Pulliam 1988), and a gene flow. Successful dispersal between forest patches is a key process for maintaining forest bird populations at the urban regional scale in the face of forest loss and fragmentation (Sutherland et al. 2000, Donnelly and Marzluff 2004).

To provide connectivity of green spaces in a hierarchy, first of all, the model needs to address the hierarchy of urban planning: neighborhood, city, and urban region. It is equally important to consider how these three scales interact with one another (Figure 4.6). To create connectivity across scales, at each scale, relevant green landscape elements can (1) be connected to each other, (2) be a part of coarser-scale elements and/or (3) be physically connected to them. For example, several streets with bioswales (i.e., vegetated swales) at the neighborhood scale can constitute a green corridor at the city scale, which, in turn, with city parks, can be a part of a regional park system at the urban regional scale (see for example, Jim and Chen 2003). Existing hierarchy of road networks (i.e., major and minor streets) can be used to develop a hierarchy of green corridors by adding street trees and stormwater swales in the center median/turn lane. Regional parks, city parks, and neighborhood (pocket) parks can be linked by these green streets. Isolated patches of habitat (e.g., small wetlands and remnant forests) can be connected using the green network such as an open stormwater drainage system and linear parks (Girling and Kellett 2005). Smaller green landscape elements such as home gardens and street trees can be connected to and be a part of larger green networks such as tree-lined streets and greenways that connect to even larger green spaces such as protected forests and regional parks (Figure 4.7).

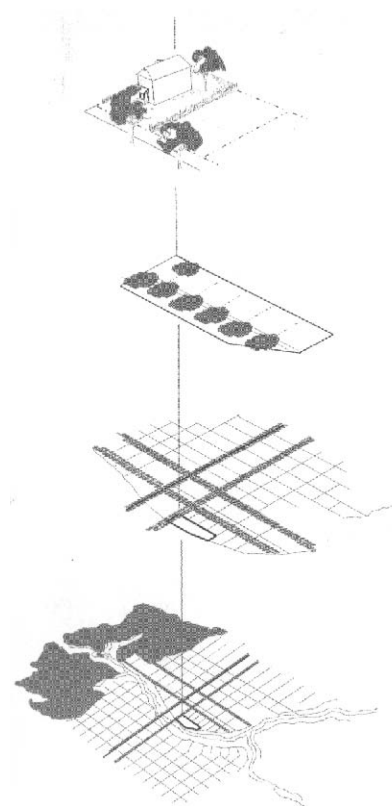

Figure 4.7: Developing connectivity of green infrastructure in a nested hierarchy. Home garden trees and street trees can constitute neighborhood-scale green infrastructure, which, in turn, composes city-wide green corridors. They, then, connect to regional green infrastructure such as large protected forests, green wedges, and regional parks (*Source*: Figure 6.1 in Girling and Kellett 2005, p.104).

4.3.2.1 At the Urban Regional Scale

The application of the meta-model to greenspace conservation planning is summarized in Figure 4.8 with the nested hierarchy of three scales: urban region, city, and neighborhood. The three scales of planning units are linked by and interact through cross-scale planning strategies and processes as well as ecological processes. At the urban regional scale, where the model is intended for an application, coarse-scale patterns matter most. The amount and spatial configuration of land uses in the urban region are important for ecological processes, the flow of water, nutrients, and organisms (Turner 1989, Forman 1995, Saura and Pascual-Hortal 2007). At this scale, coarse-scale patterns of land use/cover determine ecological processes, which, in turn, affect human land-use decisions including the amount and spatial configuration of land uses. Forman's (1995) and Forman and Collinge's (1996) indispensable patterns and the aggregate-with-outliers model are informative in planning and managing important patterns such as large patches of natural vegetation, major stream or river corridors, connecting corridors, and bits of nature throughout built and agricultural areas at the coarse scale. Forman (2008) has shown the application of these ecological models/concepts to the Greater Barcelona region, Spain. Although these models are powerful concepts, they may be too general in that they use only three land-use categories: urban, natural, and agricultural. Therefore, their direct applicability to fine-grained urban and suburban areas is limited. In the context of conservation planning, I argue that ecosystem-based (coarse-filter) conservation methods supplemented with species-specific data (target species—fine-filter) is a better model to be applied to fine-grained landscapes (see for example, Opdam et al's [2008] ecoprofile approach). Lundberg and Moberg's (2003) and Lundberg et al.'s

(2008) mobile link species approach, informing key spatial processes such as seed dispersal that are important for the production of ecosystem services and the resilience of ecosystem processes, may be a better approach in some areas.

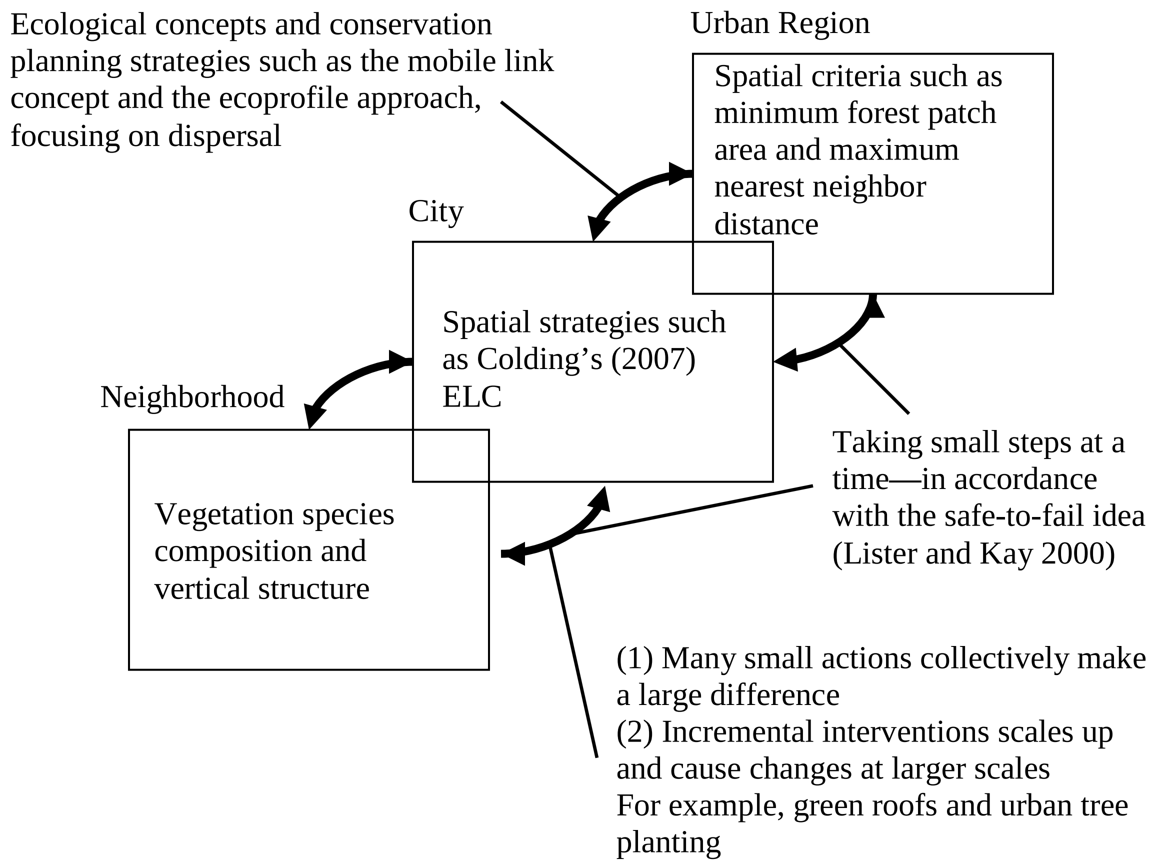

Figure 4.8: Summary of the application of the meta-model to greenspace conservation planning for the goal of creating (functional) connectivity of green spaces that can provide response/functional diversity across scales in order to conserve forest bird species and populations in an urban region.

Here I will illustrate the application of the meta-model to the landscape planning goal of conserving forest bird species at the urban regional scale. This demonstration is in accordance with my second proposition: landscape and urban planners can influence response/functional diversity and connectivity across scales to maintain, restore, or enhance the resilience of an urban region. Using the example of the conservation of forest bird species, their habitat, deciduous, some coniferous, and mixed forests can be used as the example of a focal ecosystem type. Let us further assume that “forest” cover is a reasonable surrogate for the forest ecosystem. Using a land cover map, the amount and spatial configurations of forest cover are identified for a target urban region such as the Greater Boston region. The amount of forest can be converted to the percentage of forest in a certain area such as the whole urban region, a particular district, or a sub-watershed, and can be compared with empirical data on the percentage of forest thresholds, if they exist, for maintaining the populations of forest birds. The percentage of forest can be easily monitored over time, using remote sensing data. Then, a goal to increase or maintain the regional forest cover can be set (“goal setting” in Figure 4.5) or the threshold percentage can be used in proactive planning to act before the regional forest cover percentage declines below this level (“goal setting,” “alternative future scenarios,” and “monitoring” in Figure 4.5). Also, various spatial configuration metrics of forest cover, especially the ones that have been identified to be important for predicting forest bird species abundance as a function of forest configuration (see chapter 3), can be used to assess the current state of the regional forest cover to develop goals and scenarios (“goal setting” and “alternative future scenarios”) and be monitored to see the trend of change and to see if the plan has achieved its objectives (“monitoring” and “evaluation”). The monitoring results can be used in goal setting (step 1), for example, setting a proactive % forest threshold goal and deciding on the next goals in an adaptive planning framework. The monitoring results can also be used both in scenario making as a story and to show the consequences of certain policies (step 2) and in developing a plan (step 3) based on the chosen scenario and the goals set in step 1, and most importantly in step

6, where the implemented plan is evaluated to see whether or not it has achieved the goals set or is progressing as planned (step 3). The sequence is depicted in a flowchart (Figure 4.9).

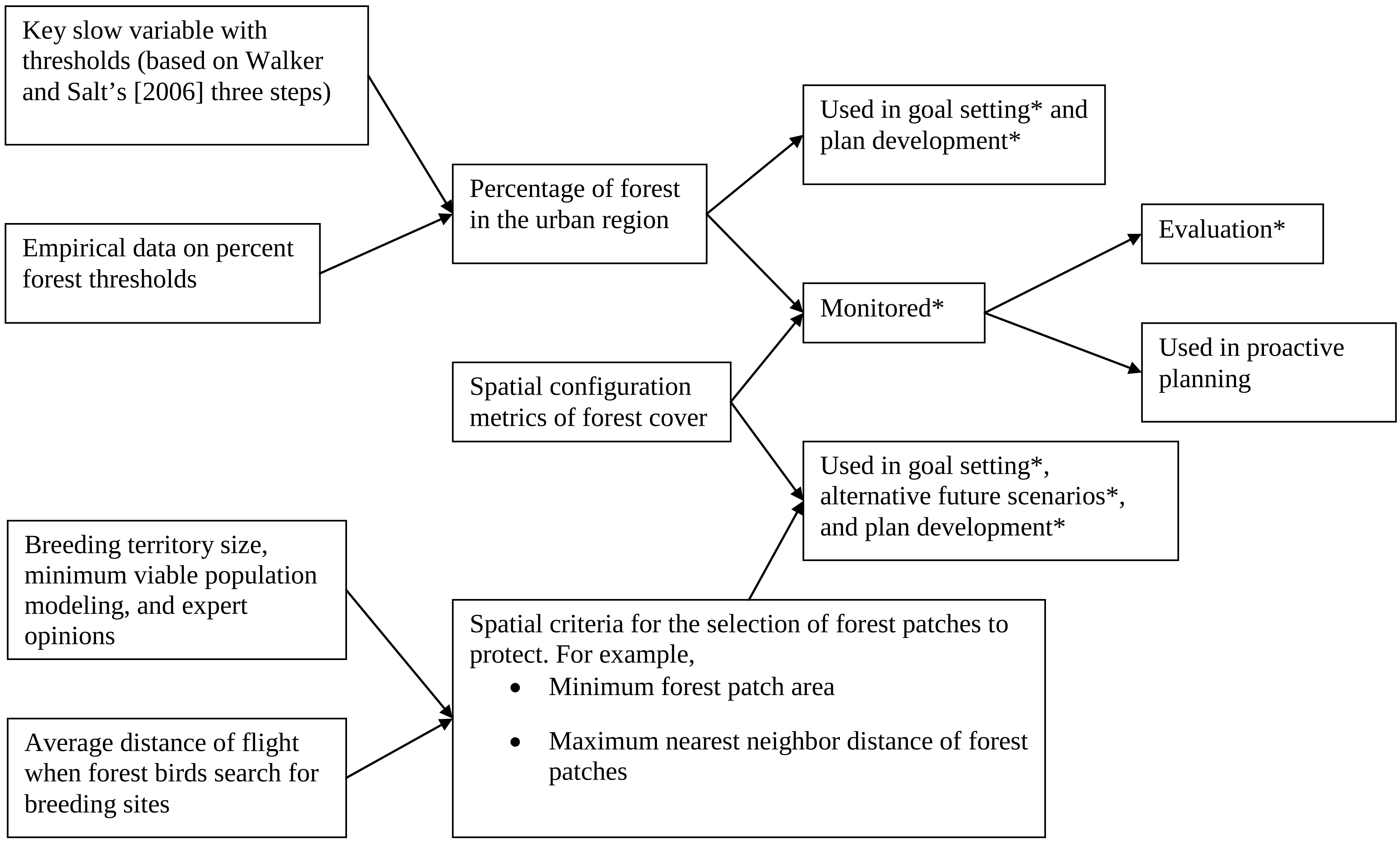

Figure 4.9: Flowchart of forest cover planning for the conservation of forest bird populations at the urban regional scale. *Refers to the steps in the landscape planning "best practice" model (Figure 4.5).

At the urban regional scale, broad patterns of forest are important for the movement and dispersal of forest birds. The percentage of forest in an urban region is equally important for area-sensitive species. Large forest patches are needed by interior species that require a certain size of core area for breeding and foraging. Spatial criteria for the selection of forest patches to protect can be established based on a breeding territory size, minimum viable population modeling, and expert opinions (see Figure 4.9). The maximum nearest neighbor distance between forest patches may be based on an average distance of flight when the forest birds search for breeding sites.

4.3.2.2 At the City Scale

At the city scale, similar spatial criteria as those at the urban regional scale can be used in deciding which forest patches to protect and where to restore the forest. The amount and spatial configuration of forest are important factors at this scale as well. The absolute amount of forest is important for some area-sensitive species; some large patches are necessary for interior species. The spatial configuration of forest is important for the movement and dispersal of forest birds. Therefore, similar spatial criteria apply to the city scale as well.

Colding's (2007) ELC is a particularly useful strategy to be applied at the city scale. It is a landscape planning strategy to provide landscape complementation and supplementation (Dunning et al. 1992) by clustering different types of open spaces (e.g., golf courses, orchards, remnant forests, cemeteries, and city parks) in a human-dominated landscape. Colding (2007) argues that clustered different types of open spaces can support emergent ecological functions, which would not be supported if specific, individual open spaces of different type were separated in a heavily urbanized matrix hostile to organisms' movement. For example, if a golf course with a pond was sited adjacent to a remnant forest, these two different types of open space could support an amphibian species to complete its life cycle, which requires a pond to lay eggs and spend the juvenile stage and upland to spend the adulthood. By clustering different types of open space, (1) the overall connected habitat area increases and (2) emergent ecosystem processes that are important for biodiversity can be supported through the mechanisms of landscape complementation and supplementation (Colding 2007).

The juxtaposition of different types of open spaces can be measured by juxtaposition indices such as the contagion index (CONTAG) at the landscape level. CONTAG can be used to analyze the current level of mixing of different land cover types in the landscape mosaic. Higher values mean that the patches of different land cover types are more aggregated and less interspersed. The isolation of different open space types as a whole can be measured by the distribution statistic of the proximity index such as PROX_AM. In terms of supporting forest bird populations, since some species require a high forest edge contrast/amount as well as certain amount of forest (see chapter 3), ELC can be used to plan thoughtful spatial configurations of different types of open spaces at the city scale.

4.3.2.3 At the Neighborhood Scale

Girling and Kellett (2005) recommend that at the neighborhood scale, functioning natural areas should be protected and, where possible, interconnected with green infrastructure at the regional scale, such as rivers and large natural areas. Isolated patches of habitat (e.g., small wetlands and remnant forests) can be connected using the green network at the neighborhood scale, such as an open stormwater drainage system and linear parks (Girling and Kellett 2005). “If designated to engage nature and natural processes, these green networks can contribute to the ecological functioning and health of the neighborhood” (Girling and Kellett 2005, p. 68).

At the neighborhood scale, different aspects of green space may be more important to support regional forest bird populations. For example, tree species composition and vertical structure may be important factors to consider (DeGraaf et al. 1998, Miller et al. 2001, Díaz et al. 2005, Lee and Rotenberry 2005) when planning neighborhood parks, street trees, and integrated open drainage systems.

Trees can also become an important component in the Sustainable Sites Initiative in terms of green infrastructure planning and assessment. The Sustainable Sites Initiative was developed in conjunction with the American Society of Landscape Architects, the Lady Bird Johnson Wildflower Center, and the United States Botanical Garden to create voluntary national guidelines and performance benchmarks for sustainable land design, construction and maintenance practices (Sustainable Sites Initiative 2009). Once finalized, the guidelines and performance benchmarks will be used to create the first rating system for sustainable landscapes, offering guidance on sustainable best practices for all sites with or without buildings (GreenInfrastructure.net 2009, Sustainable Sites Initiative 2009). The Initiative will work as a stand-alone rating system, and also be incorporated into the LEED® (Leadership in Energy and Environmental Design) Green Building Rating System™ by 2011 (American Society of Landscape Architects 2009). The proposed rating system is currently being tested, using 75 to 150 diverse projects to see how well the rating system applies to design construction and maintenance practices (GreenInfrastructure.net 2009, Sustainable Sites Initiative 2009). Trees can contribute to making a site protect, restore, and regenerate ecosystem services. For example, trees and maintained shrub layers to add vertical depth are not only aesthetically pleasing but also contribute to cleaning air, increasing water retention and evapotranspiration, having more areas for water to permeate into the soil, and may even have some habitat value.

4.3.2.4 Processes that Connect Different Scales

The focus of the meta-model is on the processes that link planning units and planning processes across space and time. In the context of the conservation of forest bird populations in an urban region, the cumulative effects of green roofs and tree planting are an example of these processes that start out at the neighborhood scale and cascade up to the city scale and beyond (see Figure 4.8). With regard to green roofs, their effects such as reduction in stormwater runoff and the urban heat island effect may be limited at a specific site but the cumulative effects of many green roofs extend to the city scale and beyond. For example, the study conducted as a precursor to the City of Toronto's green roof bylaw calculated the potential monetary savings of citywide green roofs to be $313,100,000 initially and $37,130,000 annually from the combined benefits of stormwater, combined sewer overflow, air quality, building energy, and urban heat island (Ryerson University 2005). Citywide green roof initiatives can be seen in Chicago and Toronto, for example (Department of Environment, City of Chicago, 2009, City of Toronto 2009).

Another example of the processes that cascade across spatial scales is tree planting. Although each tree's effect is locally limited, if enough number of trees are planted, they can have large cumulative citywide effects. Cities such as Baltimore and New York have initiated urban tree planting programs (City of Baltimore 2009, MillionTreesNYC 2009). Urban tree planting has been reported to have many benefits such as the development of social capital, reduction of crimes, and environmental education beyond environmental benefits (Dwyer et al. 1992, McPherson 1992, McPherson et al. 1994, Kuo and Sullivan 2001, Westphal 2003). Urban tree planting can also be a timely climate change mitigation strategy because of carbon storage and sequestration potential of urban trees (Nowak and Crane 2002, Kollin and Schwab 2009). It can also be a climate change adaptation strategy by reducing urban heat island effects (McPherson and Rowntree 1993, McPherson et al. 1997).

The processes whereby (1) many small effects at the neighborhood scale collectively make a large impact at the city scale and (2) incremental interventions at a fine scale scale up and cause changes at larger scales are similar to bottom-up, grassroots activities in creating social changes. Citizen groups, neighborhood associations, and grassroots organizations can initiate projects that individually may be considered trivial but collectively can combine in physical or functional aspects to make a large impact to the whole city. Also, one neighborhood's new idea or project, if it is successful, can be imitated by other neighborhoods, and soon the entire city can experience the benefits. The way successful projects spread to a higher jurisdictional/planning level is the same for other levels. For example, successful projects at the city level can spread to the urban regional level. This process has a temporal dimension as well. Like movies and other products, a great "buzz" of the project can create a new identity, civic pride, or long-term success.

In open-space planning, Erickson (2006) describes Vancouver's neighborhood greenway projects where a successful neighborhood project can spread to other neighborhoods. This strategy of taking small steps can work at the city scale as well. For example, one city's greenway corridor project can be imitated by other cities and they can build on to one another, creating a regional greenway network. Taking a small step at a time is also in accordance with the safe-to-fail idea (Lister and Kay 2000). Small demonstration projects generally have smaller negative impacts even if they fail, allowing decision-makers to take the risk. With the integration of testable hypotheses into the new design and monitoring into the planning process and the budget, planners and social and biological scientists can still learn from the projects even if they fail. Actually, the mindset that turns controlled failures and surprises into opportunities rather than liability or failures to predict is critical for successful adaptive planning (Lee 1993, Lister and Kay 2000, Kato and Ahern 2008). When they are successful, small demonstration projects can be adopted by other neighborhoods and cities as people feel more comfortable with new ideas (Erickson 2006).

As for ecological processes that connect across scales, I am focusing on the function of dispersal because successful dispersal between forest patches is a key process for maintaining forest bird populations at the urban regional scale in the face of forest loss and fragmentation (Sutherland et al. 2000, Donnelly and Marzluff 2004). Ecological concepts and conservation planning strategies such as the mobile link concept and the ecoprofile approach use specific processes and associated species to manage and plan for the processes that are important for maintaining ecosystem services. The mobile link concept is a species-based approach but the species are linked to specific spatial processes (Lundberg et al. 2008). The mobile link species therefore represent certain ecosystem processes such as seed dispersal and pollination that "link" (1) spatially separated resource patches of the same kind and (2) different types of patches (Lundberg and Moberg 2003, Lundberg et al. 2008). Because ecosystem processes do not recognize administrative boundaries and often operate between the city and the urban regional scales, the specific spatial processes of the mobile link concept are examples of those processes that link across scales in the meta-model (Figures 4.6 and 4.8). The ecoprofile approach (Opdam et al. 2008) specifically uses species dispersal in its matrix for each ecosystem type (e.g., forest, grassland, wetland, etc.) in developing an ecosystem network. For each ecosystem type, the ecoprofile of certain species is defined by dispersal capacity (i.e., the maximum inter-patch distance that can be crossed during dispersal) and the minimum ecosystem area requirement for the species' population survival in the network. The ecoprofile approach can be used to plan an ecosystem network for each ecosystem type targeted for a suite of species at the regional scale. Dispersal distance can be used in the negotiation process to plan ecosystem networks with various levels of investment. In other words, dispersal can be used to scale ecosystem networks and to adjust to various aspiration levels (Opdam et al. 2008). For example, when the ecoprofile method was applied to develop animal movement corridors, relatively mobile species such as otter would require the least investment, whereas the species with shorter dispersal capacity would require more investment to develop corridors more densely and in shorter distances in between (Opdam et al. 2008). This is how dispersal is used to relate to different scales.

4.4 Conclusions

The main research question is: How can planning and design cultivate or improve the capacity of an urban region to provide ecosystem services over time in the context of change? I have argued that response/functional diversity, redundancy, and connectivity across scales are key to the resilience of a social-ecological system and the sustained provision of ecosystem processes and services. Although there is a growing recognition that these ecological concepts are key to the maintenance of ecosystem functions over time, a link to planning and design application has not been strongly established. Then the question becomes: How can the concepts of response and functional diversity, redundancy, and connectivity across scales be translated to landscape planning and design—specifically, greenspace conservation planning in urban regions? I argue that response/functional diversity and cross-scale connectivity are the aspects of a social-ecological system that planning and design can intentionally create, protect, or restore in a conservation planning framework, and that enable it to maintain its resilience. To increase response diversity, a variety of green spaces with different amounts and configurations of forest cover should be provided that behave differently to change and disturbance. To create connectivity of green spaces across scales, at each scale, relevant green landscape elements can (1) be connected to each other, (2) be a part of coarser-scale elements and/or (3) be physically connected to them. For example, existing hierarchy of road networks (i.e., major and minor streets) can be used to develop a hierarchy of green corridors. An important caveat here is that over-connected systems are susceptible to shocks; undesirable disturbances such as disease and pest outbreaks are easily transmitted through the system. Therefore, due caution needs to be exercised when deciding where to protect, restore, and create green space, and the decision needs to be based on the evaluation of its relative importance to the target ecological process and against costs.

I have then proposed a meta-model to show the importance of cross-scale interactions, as argued based on the panarchy concept (Gunderson and Holling 2002), in an organizing framework that links at least three scales: neighborhood, city, and region. The meta-model serves as an organizing framework for various planning concepts and strategies by placing them in hierarchical dynamics, and helps planners to see the connection between them (see Figure 4.6). The meta-model answers the need by urban and regional planners for a conceptual framework to work with, which would show them how each scale is related to one another in a nested hierarchy and what tools or concepts can be used to link one scale to another.

Then, I have demonstrated the application of the meta-model to greenspace conservation planning for the purpose of conserving forest bird species in urban regions (Figure 4.8). In greenspace conservation planning applications, the focus of the meta-model is on the processes that "connect" physically separated patches and patches of different types as well as physically connected patches both across and within scales. The model connects target scale dynamics to the scales above and below—cross-scale interactions. At the urban regional scale, broad patterns of forest are important for the movement and dispersal of forest birds. Also, the percentage of forest in an urban region is important for area-sensitive species. Large forest patches are needed by interior species that require a certain size of core area for breeding and foraging. To operationalize the model, spatial criteria for the selection of forest patches to protect can be established based on a breeding territory size, minimum viable population modeling, and expert opinions (Figure 4.9). The maximum nearest neighbor distance between forest patches may be based on an average distance of flight when the forest birds search for breeding sites. At the city scale, similar spatial criteria as those at the urban regional scale can be used in deciding which forest patches to protect and where to restore the forest. Colding's (2007) ELC is a particularly useful strategy to be applied at the city scale. At the neighborhood scale, different aspects of green space, such as tree species composition and vertical structure, may be important factors to consider when planning neighborhood parks, street trees, and integrated open drainage systems.

The focus of the meta-model is on the processes that link planning units and planning processes across space and time. In the context of the conservation of forest bird populations in an urban region, the cumulative effects of green roofs and tree planting are an example of these processes that start out at the neighborhood scale and cascade up to the city scale and beyond. With regard to green roofs, their effects such as reduction in stormwater runoff and urban heat island effects may be limited at a specific site but the cumulative effects of many green roofs extend to the city scale and beyond, and these benefits can be calculated in monetary terms at the city scale. Similarly, if enough number of trees were planted, they could have large cumulative citywide effects. Urban tree planting can also be a climate change mitigation and adaptation strategy. In sum, green roofs and urban tree planting are examples of cross-scale interactions where (1) many small actions collectively make a large difference and (2) incremental interventions scale up and cause changes at larger scales (Figure 4.8). Taking a small step at a time—for instance, a successful neighborhood project spreading to other neighborhoods—is also in accordance with the safe-to-fail idea (Lister and Kay 2000). Small demonstration projects generally have smaller negative impacts even if they fail, allowing decision-makers to take the risk. With the integration of testable hypotheses into the new design and monitoring into the planning process and the budget, planners and social and biological scientists can still learn from the projects even if they fail. Finally, ecological concepts and conservation planning strategies such as the mobile link concept (Lundberg and Moberg 2003, Lundberg et al. 2008) and the ecoprofile approach (Opdam et al. 2008) are examples of the processes, such as dispersal, that often operate between the city and the urban regional scales (see Figure 4.8). These concepts and approaches use specific processes and associated species to manage and plan for the processes that are important for maintaining ecosystem services.

The focus of the research is the spatial configuration of land use, especially, green spaces, in urban regions—how green spaces can be best configured to increase the resilience of an urban region by applying ecological concepts such as response/functional diversity, redundancy, and connectivity across scales. The related planning concepts and strategies described are ways to deal with inevitable surprises and disturbances and to increase the resilience of a landscape. Variability in the landscape is the key to maintaining renewal capacity when the landscape undergoes some change (Holling et al. 2002b). Planning and design should develop landscapes that are (1) more spatially heterogeneous—not fragmented but integrated at multiple scales—by adding fine-scaled elements such as hedgerows, wind breaks, pockets of restored nature, diverse crops, integration and preservation of cultural/historical heritage (stone walls, monuments), etc. and (2) more functionally diverse (i.e., multifunctional landscapes) across and within scales. By so doing, response and functional diversity will be increased, making a landscape less sensitive to disturbances and building its capacity for enhanced resilience (Holling and Gunderson 2002).

Table 4.1: At each scale, examples of green spaces which include trees and shrubs are identified. They can contribute broadly to ecosystem processes and services; narrowly, they can support forest birds in the context of forest bird species conservation. Different levels of biological organization are matched up with relevant planning scales to manage and plan for them. *Landscape and urban planning is considered at three nested scales: region (37.5 x 37.5 km or 60 x 60 miles), city (up to 9,308 hectares or 23,000 acres), and neighborhood (51 to 202 hectares, or 125 to 500 acres) (Girling and Kellett 2005, Sipes and Lindhult 2007).

Scale*	Green Spaces	Levels of Biological Organization
Urban Region	Connected system of regional parks (e.g., the Emerald Necklace), greenways, riparian vegetation, nature reserves, protected forests, stepping-stone habitats, corridors	Ecosystems, biological communities, species, metapopulations
City/town, community	Local parks, green infrastructure, smaller conservation area, corridors, orchards, cemeteries, schoolyards, community gardens, golf courses, remnant forests, street trees, rooftop gardens	Local populations (subpopulations)
Neighborhood, site	Neighborhood parks, street trees, bioswales, house gardens, rooftop gardens	Subpopulations, genes

CHAPTER 5

CONCLUSION

5.1 Introduction

With rapidly declining biodiversity around the world, biodiversity conservation should, arguably, be identified as one of the major goals in urban regional planning. An urban region, or a city-region (i.e., a city and its underlying suburbs), is highly heterogeneous with complex, multidirectional, continuous and dynamic processes (e.g., land development, land abandonment, forest loss, forest regeneration, population increase or decrease, water and species movement). An urban region is a complex adaptive system (Lansing 2003, Levin 2003, Norberg and Cumming 2008). Urban regions are also where most people live in the U.S. (Hobbs et al. 2002) and in many parts of the world (United Nations 2008), and often coincide with the areas of high biodiversity conservation priority (Groves et al. 2000, Balmford et al. 2001, Araújo 2003). An urban region is also a relevant scale for conservation planning/design/management, especially for species such as forest birds that have a large home range and a long dispersal distance. Therefore, I have argued that land-use plans for urban regions need to explicitly integrate the conservation of biodiversity as a recognized priority (Ahern et al. 2006). Ecological data collected in urban regions for biodiversity conservation should be used in urban regional planning to develop an environment where humans and non-human species can co-exist in harmony. Applying landscape ecological theories and principles such as complementation/supplementation, spatial heterogeneity, and connectivity to landscape planning would help develop such an environment, and I have contributed to this end by proposing a greenspace conservation planning framework that has integrated some key ecological concepts for increasing the capacity for resilience.

In this dissertation, I have asked the question of how landscape ecological planning can be advanced by incorporating relevant ecological data and ecological theories and concepts to better plan urban regions for increased resilience, ecosystem services, and ultimately, for sustainability. A gap still exists between the knowledge of specific ecological process and the expressed spatial land use patterns resulting now mostly from human activities, given the constraints of the abiotic environment (e.g., steep slopes, bedrock geology, water bodies, etc.) (Opdam et al. 2001). The public and researchers are becoming increasingly aware of the unsustainable practices of land use, such as suburban sprawl and highly subsidized highway systems, but often do not know how to act to correct them (Berke 2009). With regard to the divide between the accumulated knowledge of conservation biology and land use planning, the knowledge has not been effectively put into practice to "determine where to act, what to conserve, and how to create strategies that support green design in local land use planning" (Berke 2009). I echo Ndubisi's (2002a) concern that there are not enough procedural methods to integrate best available landscape ecological and conservation biological knowledge into effective land use planning to achieve ecological sustainability. My research has addressed this knowledge/procedural gap by demonstrating how available ecological data can be used in a greenspace conservation framework through the proposed meta-model and its application to greenspace conservation planning. In chapter 3, an empirical study is conducted on the relationship between forest bird abundance and landscape characteristics, focusing on the composition and configuration of forest cover in metropolitan regions of the eastern U.S. Forest birds are treated as a species-level example of biodiversity and forest cover is treated as a key land use (cover) type that can be targeted and managed for planning. Drawing on the results from this cross-scale observational study, in chapter 4, a meta-model for urban and landscape planning and design is developed with the landscape planning best practice model, and the application of the meta-model to greenspace conservation planning for the purpose of conserving forest bird populations at the urban regional scale has been demonstrated. In this concluding chapter, I will examine the research hypotheses and questions in chapter 3, summarize and discuss the main points of chapter 4, and evaluate the research propositions in chapter 4. I will then discuss their broader implications for landscape planning of an urban region and make some planning recommendations. I will end with the discussion on the directions of future research.

5.2 Examination of the Research Hypotheses and Questions in Chapter 3: Route-level, Multi-scale Analysis of Forest Bird Abundance-Habitat Relationships in Urban Regions across the Eastern United States

The hypothesis of the forest bird-habitat relationship study (chapter 3) is: the percentage of forest cover and the connectivity of forest cover in the surrounding landscape are positively correlated with the number of individuals of the selected forest bird species. The results of the simple linear regression of bird abundance against the percentage of forest cover showed that the hypothesis that bird abundance is positively correlated with the percentage of forest cover in the surrounding landscape is correct (Figures 3.7, 3.11, 3.15). Bird abundance increased as the percentage of forest cover in the landscape increased. The regression slope estimates ranged between 0.009 and 0.046 for the square root transformed bird abundance across scales (Tables 3.6, 3.10, 3.14).

Within this hypothesis, there is a research question relating to thresholds: Do the selected birds exhibit a threshold response to the percentage of forest cover in a landscape? Notwithstanding the "noisy" data, all species at all scales except for Ovenbird (OVEN) at the 6 km buffer size showed a threshold response. Stronger and more stable thresholds were identified for Black-and-white Warbler (BWWA) at 87% forest cover at the 180 m buffer size (Figure 3.9, Table 3.8) and at 86% forest cover at the 6 km buffer size (Figure 3.10, Table 3.16) and for Wood Thrush (WOTH) at 9% forest cover at the 180 m buffer size (Figure 3.11, Table 3.8) by the one-breakpoint piecewise linear regression models. Because the adjusted R^2 value for WOTH is still very low at 0.07 and high impervious cover is a better predictor of its abundance than forest cover, the threshold has little planning and management significance of forest cover for WOTH. Because nearly identical threshold forest cover percentages are identified for two out of the three scales for BWWA, 87% seems to be a persistent threshold. However, maintaining an average forest cover in an urban region above this threshold value of 87% would be unrealistic. Instead, based on the data analysis conducted, I would recommend protecting large forest patches—for example, large enough to contain an interior area for a breeding territory—and maintaining their connectivity in the urban region because BWWA is a forest interior species (Whitcomb et al. 1981, Fahrig 1999) with very low tolerance to fragmentation (Whitcomb et al. 1981). Forest cover connectivity can be measured by connectivity metrics such as COHESION, CONNECT, GYRTAE_AM, ENN_AM, PROX_AM, and SIMI_AM at the forest cover class level (McGarigal et al.

2002). The existence of thresholds can be used as a basis of support for proactive planning, taking actions before the amount of forest in an urban region is reduced below the threshold level or it can serve as a useful target of restoration. This would translates to conservation planning actions such as prioritizing land management or acquisition options, and targeting areas for restoration. (See section 5.5.3 for another application of threshold-based planning to urban watersheds.) The obstacles to threshold-based planning include a lack of species-specific data, difficulty in detecting thresholds, and the danger of over-simplifying complex social-ecological systems.

Related to the correlation between bird abundance and forest cover, another question is: What land cover type including forest cover is the best predictor of the forest bird abundance? The research found that forest cover was not always best correlated with bird abundance. Other land cover types were better correlated with bird abundance for Eastern Wood-Pewee (EAWP) and WOTH. For EAWP, other land cover types such as open space, low imperviousness, and high imperviousness explained much more variation in bird abundance than forest cover type. For WOTH, either shrub cover or high imperviousness explained the most variation in bird abundance and forest cover was the second. The results are corroborated by the very low R^2 values for EAWP and WOTH in the simple linear regression models. The results also mean that for the conservation of EAWP and WOTH the percentage of forest cover in a landscape is not the most important factor as other land cover types are better predictors of their abundance. Moreover, the important variables in the reduced models for these species show that the diversity of land cover types (-), contrast-weighted forest edge density (+), and contagion (+) are important across scales. (The sign in the parenthesis indicates the sign of the variable's partial regression coefficient.) Therefore, the planning and management significance of these variables is that for these species which are more fragmentation-tolerant and can use the edge as well as the interior of forest patches (Whitcomb et al. 1981), a landscape should contain fewer land cover types with less even proportion and more forest edge density and/or edge contrast, and the patches in the landscape become more aggregated (i.e., more like-cell adjacencies) and less interspersed (i.e., inequitable distribution of pairwise adjacencies). In other words, overall, EAWP and WOTH favors a landscape composed of aggregated patches with little land cover diversity, and these patches to have a high contrast to forest cover.

Another study question is: Do important forest composition and spatial configuration factors vary when measured at different spatial scales? This question is examined together with the other part of the initial hypothesis: the connectivity of forest cover in the surrounding landscape is positively correlated with bird abundance. As connectivity is a complex, and an emergent, concept affected by the interaction between landscape structure and the particular ecological process of interest (Green 1993, Tischendorf and Fahrig 2000a, b, Turner et al. 2001), there is no one measure or landscape metric of connectivity (McGarigal et al. 2002). Functional connectivity is especially difficult to measure and be expressed by landscapes metrics. The landscape metrics that represent an aspect of connectivity used in the additive, full multiple regression model are COHESION, CONNECT, GYRTAE_AM, ENN_AM, PROX_AM, and SIMI_AM. They are mostly measures of structural connectivity or isolation. SIMI_AM represents functional connectivity to some extent based on the landscape mosaic perspective (Wiens et al. 1993, Ricketts 2001, Haila 2002, McGarigal et al. 2002, Bender and Fahrig 2005).

The hypothesis, the connectivity of forest cover in the surrounding landscape is positively correlated with bird abundance, held true except for one connectivity metric, PROX_AM (-), at the 180 m buffer size for EAWP. The other connectivity metrics such as SIMI_AM, CONNECT, GYRATE_AM, and COHESION, identified as important variables in the reduced models, were all positively correlated with bird abundance at multiple scales. SIMI_AM in particular was most often selected as the important variable across scales. This means that for the selected forest bird species, the landscape consisted of land cover types that are similar to forest cover type (in terms of the average percent tree canopy) is a hospitable environment—appearing functionally connected.

For each species, although there was some variation in the important forest composition and spatial configuration factors at different spatial scales, some variables were consistently identified as important across scales. For example, PLAND (+) was important for BWWA and OVEN at three scales and for American Redstart (AMRE) at two scales. SIDI (-) was important for WOTH at three scales. SIMI_AM (+) was important for OVEN at two scales, so were CONTAG (+) and CWED (+) for EAWP. CWED (-) was important for BWWA at two scales. The results indicate that OVEN requires a highly connected forest cover as does BWWA with low forest edge density and/or contrast across scales. AMRE requires a high forest cover in a broader area with less forest patch density and less forest edge density and/or contrast, but prefers more complex forest patch shapes in a small area. WOTH consistently requires little land cover diversity and its requirement for a small area is mixed with high forest cover connectivity but high forest edge density and/or contrast. EAWP, for a broad landscape, requires patches of various land cover types to be more aggregated and less interspersed but high forest edge density and/or contrast. EAWP's requirement for a small area is the opposite: it favors patches belonging to the land cover types that are similar to forest cover type to be less aggregated and more interspersed, and forest patches to be more isolated, but low forest edge density and/or contrast. In other words, for a small area, EAWP favors forest patches to be more isolated and fragmented but forest patch shapes to be more compact, frequently embedded in the area where existing land cover types that are similar to forest cover (thus, less forest edge contrast).

Planning implications for these findings are that (1) there is variability for habitat preference even among species that share similar life history characteristics (i.e., neotropical migrant, forest-breeding birds), and that these species-specific requirements should be provided in a broader management framework based on the most common variables, i.e., PLAND (+), CWED (+) and (-), and SIMI_AM (+) and (2) important variables such as SIMI_AM, CONTAG, and CWED can be addressed by spatial concepts such as Colding's (2007) ecological land-use complementation, which proposes to aggregate various greenspaces, thereby increasing the total habitat area and functional connectivity. The percentage of forest cover in a landscape is the most important variable; however, it is not just forest but forest and other land uses in some combination of proximity and spatial arrangement that are also important for the conservation of the forest bird species.

There is yet another question: What would be a reasonable urban forest cover goal to support the selected forest birds in urban regions across the eastern U.S.? The stronger thresholds identified by the one-breakpoint models are about 87% forest cover for BWWA and 9% for WOTH. These values seem either too high or too low to be realistic in terms of managing regional urban forest cover. Moreover, BWWA, WOTH, and AMRE might decrease in number when forest cover is too high, above 87% (Figures 3.9, 3.12, 3.13, 3.14, 3.15). American Forests recommend an average tree canopy cover of 40 percent of the land area for cities east of the Mississippi and in the Pacific Northwest (American Forests 2010). For downtown areas, they recommend 15 percent cover; for urban residential areas, 25 percent cover; and for suburban residential areas, 50 percent cover (American Forests 2010). These percentage values are meant to be general goal guidelines to achieve environmental and quality of life goals, including federal and local clean air and water regulations (American Forests 2010). In the end, each community must set its own tree canopy cover goals (American Forests 2010). The general goal guidelines (i.e., 15-50% forest cover) do coincide with the lower stable thresholds identified by the two-breakpoint piecewise regression models. For example, 24% and 86% forest cover at the 180 m buffer size for BWWA (Figure 3.14) and 36% and 71% forest cover at the 180 m buffer size for AMRE (Figure 3.15, Table 3.8). Because unit increase in % forest contributes more to the increase in bird abundance over the lower threshold, it pays to keep an average forest cover above these lower thresholds by protecting and restoring forests. These forest cover goals can be achieved by urban tree planting programs, for example, which seem to have become popular in some U.S. cities (City of Baltimore 2009, City of Boston 2009, MillionTreesNYC 2009), as a way to combine environmental benefits with aesthetics, environmental education, urban biodiversity, social justice, and climate change mitigation and adaptation strategies

(Dwyer et al. 1992, McPherson 1992, McPherson and Rowntree 1993, McPherson et al. 1997, Kuo and Sullivan 2001, Nowak and Crane 2002, Westphal 2003, Kollin and Schwab 2009). The city-scale tree planting efforts could be related to the increase in regional tree cover by (1) a coordination with regional-scale greenspace conservation programs such as an urban growth boundary, the protection of regionally important, large tracts of forests such as groundwater recharge areas, riparian forests, and protected forests and (2) by the spread of tree planting movements to neighboring towns and cities as an example of social processes that gain popularity and momentum and cascade up scales (see Figure 4.8).

Finally, the questions of (1) landscape ecological planning concepts and strategies that can be applied to protect or restore the amount and spatial configuration of forest patches that support the selected forest bird species, (2) "collateral" functions and benefits (e.g., recreation, water quality, and cultural landscape protection) that can be reasonably associated with the given amount and spatial configuration of forest cover, and (3) the issue of scale and how it affects different scales of planning were discussed in chapter 2 and chapter 4 in the development of the greenspace conservation planning framework for urban regions, and important points will be reiterated and further discussed throughout the subsequent sections.

5.3. Chapter 4, "Development of a Landscape Planning Meta-model and its Application to Greenspace Conservation Planning in Urban Regions based on the Resilience Concept," Summary and Discussion

First, building on existing general and comprehensive landscape planning models (i.e., Steinitz 1990, Steiner 1991, Ahern 1999, and Leitão 2001), especially, Ahern's

(1999) framework model, I have developed a landscape planning “best practice” model (Figure 4.5). It shows cyclic and iterative steps of landscape planning with feedback loops. Its characteristic is the explicit integration of monitoring into the planning steps to learn by doing. The best practice model can be applicable to any scale to achieve various planning goals.

Second, building on the understanding of resilience thinking, especially, adaptive cycles in a linked nested hierarchy over spatial and temporal scales (i.e., panarchy), I have developed a general meta-model for landscape planning, showing interactions across scales (Figure 4.6). The meta-model presents the interconnections between different landscape planning scales in a nested hierarchy, incorporating the landscape planning best practice model at each scale. The meta-model presents a common framework which addresses the interconnections across landscape planning scales from urban region, to city, and to neighborhood. The meta-model is intended to help planners recognize cross-scale processes that are operating, organize their thinking on these processes, and identify the processes that are particularly important to their planning decision, plan or project at hand. Then, applying this meta-model, I have developed a conceptual framework for greenspace conservation planning in urban regions (Figure 4.8), using the results of the forest bird-habitat study (chapter 3). I believe that the meta-model’s application to biodiversity conservation planning in urban regions demonstrates its usefulness as a tool for landscape and urban planners to identify and recognize important cross-scale processes (e.g., social and ecological processes) as well as relevant processes and dynamics operating at the scale of primary concern.

Within the application of the meta-model to greenspace conservation planning, I have shown the ways to create (functional) connectivity of green spaces that can provide response/functional diversity within and across scales for the purpose of conserving regional forest bird populations. In landscape planning for conservation of biodiversity, the concept of functional diversity can translate to, for example, replicating connectivity across and within scales (e.g., Li et al. 2005a). Creating a linked system of connectivity across scales is a good practice to increase resilience just as a linked network of linear open spaces such as greenways are a good example of a connected network for recreational and/or hydrological functions (Fábos 2004). To create connectivity of green spaces across scales, at each scale, relevant green landscape elements should (1) be connected to each other, (2) be a part of coarser-scale elements and/or (3) be physically connected to them. For example, the existing hierarchy of road networks (i.e., major and minor streets) can be used to develop a hierarchy of green-street corridors. Since over-connected systems are susceptible to shocks and undesirable disturbances such as disease and pest outbreaks are easily transmitted through the system, due caution needs to be exercised when deciding where to protect, restore, and create green space, and the decision needs to be based on the evaluation of its relative importance to the target ecological process and against costs (Simberloff and Cox 1987, Rosenberg et al. 1997, Beier and Noss 1998, Bennett 1999). It is, therefore, important to monitor the use and/or the flow of organisms, nutrients, and water through the established connectivity, and make this monitoring component be an explicit part of the planning process and budget, so that we can evaluate the effectiveness of established corridors, for example, and adapt to the findings.

In greenspace conservation planning applications, the focus of the meta-model is on the processes that “connect” physically separated patches and patches of different types as well as physically connected patches both across and within scales (see Figure 4.8). The model connects target scale dynamics to the scales above and below—cross-scale interactions. At the urban regional scale, broad patterns of forest are important for the movement and dispersal of forest birds. Also, the percentage of forest in an urban region is important for area-sensitive species such as Ovenbird and Black-and-white Warbler (Whitcomb et al. 1981, Robbins et al. 1989, Lee et al. 2002). Large forest patches are needed by interior species, such as Ovenbird, Black-and-white Warbler, and American Redstart (Whitcomb et al. 1981, Fahrig 1999), which require a certain size of core area for breeding and foraging. To operationalize the model, spatial criteria for the selection of forest patches to protect can be established based on a breeding territory size, minimum viable population modeling, and expert opinions (Figure 4.9). The maximum nearest neighbor distance between forest patches may be based on an average distance of flight when the forest birds search for breeding sites. At the city scale, similar spatial criteria as those at the urban regional scale can be used in deciding which forest patches to protect and where to restore the forest. Colding’s (2007) ecological land-use complementation is a particularly useful strategy to be applied at the city scale. At the neighborhood scale, different aspects of green space, such as tree species composition and vertical structure, may be important factors to consider when planning neighborhood parks, street trees, and integrated open drainage systems.

The focus of the meta-model is on the processes that link planning units and planning processes across space and time. In the context of the conservation of forest bird populations in an urban region, the cumulative effects of green roofs and tree planting are an example of these processes that start out at the neighborhood scale and cascade up to the city scale and beyond (Figure 4.8). With regard to green roofs, their effects such as reduction in stormwater runoff and urban heat island effects may be limited at a specific site but the cumulative effects of many green roofs extend to the city scale and beyond, and these benefits can be calculated in monetary terms at the city scale. Similarly, if enough number of trees were planted, they could have large cumulative citywide effects. Urban tree planting can also be a climate change mitigation and adaptation strategy. In sum, green roofs and urban tree planting are examples of cross-scale interactions where (1) many small actions collectively can make a large difference and (2) incremental interventions have the potential to scale up and cause changes at larger scales. Taking a small step at a time—for instance, a successful neighborhood project spreading to other neighborhoods—is also in accordance with the "safe-to-fail" idea (Lister and Kay 2000). Small demonstration projects generally have smaller negative impacts even if they fail, allowing decision-makers to take the risk. With the integration of testable hypotheses into the new design and monitoring into the planning process and the budget, planners and social and biological scientists can still learn from the projects even if they fail. Finally, ecological concepts and conservation planning strategies such as the mobile link concept (Lundberg and Moberg 2003, Lundberg et al. 2008) and the ecoprofile approach (Opdam et al. 2008) are examples of the ecological processes, such as dispersal, that often operate between the city and the urban regional scales (see Figure 4.8). These concepts and approaches use specific processes and associated species to manage and plan for the processes that are important for maintaining ecosystem services and therefore, can be recommended for integration with planning.

5.4 Evaluation of the Research Propositions in Chapter 4

Chapter 4 posed two research propositions, which were supported in the chapter. Here, I will summarize the argument in support of the propositions. The first research proposition is that response/functional diversity, redundancy, and connectivity across scales are key to the resilience of a social-ecological system. The second research proposition is that landscape planners and designers can develop, maintain, or restore these attributes by influencing land use patterns and regional development and growth.

Walker and Salt (2006) proposed three steps to manage for and enhance resilience of a social-ecological system: step 1, to understand the drivers (i.e., slow, controlling, coarse-scale variables often coupled with fine-scale, fast variables); step 2, to know the thresholds on the drivers; and step3, to enhance aspects of the system that enable it to maintain its resilience. To address the last step by landscape or urban planning and design, it can be broken down into two sub-steps. The first is to identify these aspects and the second is to develop a plan, scheme, or strategy to enhance the aspects by planning and design. The first proposition corresponds to the first sub-step: response/functional diversity, redundancy, and connectivity across scales are these attributes of a social-ecological system that are essential to build resilience capacity. Response diversity provides a “buffer” for a lost or altered function in the face of disturbance and over time during the reorganization phase because even when one species declines or becomes extinct, if there are other species that perform the same/similar function and have different sensitivity to a particular disturbance, this ecological function is more likely to be maintained, leading to the resilience of the ecological function (Daily 1997, Peterson et al. 1998, Gunderson et al. 2002b, Holling et al. 2002b, Hooper et al. 2005, Elmqvist et al. 2003, Walker and Salt 2006). Functional diversity is both within- and between-scale diversity, which produces an overlapping reinforcement of function that is remarkably robust (Peterson et al. 1998, Walker et al. 1999, Gunderson and Pritchard 2002, Gunderson et al. 2002b, Holling et al. 2002b). Redundancy reinforces response and functional diversity (Holling et al. 1995, Walker and Salt 2006). Therefore, redundancy of landscape elements at each scale and across scales is a way to increase resilience with the trade-offs of increased maintenance cost and cost to restore/develop these elements. Connectivity across scales can provide a wide and comprehensive coverage by the network; and this is an efficient coverage because of the integration of multiple scales—a finer scale for a small area and a coarse scale for a large area. I have argued that by explicitly relating these important concepts to landscape planning, especially in the context of biodiversity conservation, the resilience of an urban region can be enhanced.

The second research proposition is supported by the argument that green spaces in an urban region can be configured in such a way to increase the resilience of an urban region by applying ecological concepts such as response/functional diversity, redundancy, and connectivity across scales. In the context of biodiversity conservation planning, response diversity translates to having many landscape components, such as stepping-stone habitats, bioswales, corridors, rooftop gardens, and protected forests, which perform the same function of providing habitat for some target species. Although there may be redundancy in the functions (e.g., air and water purification, pollutant removal, water retention, and microclimate amelioration) provided, these landscape elements or green spaces would respond differently to change and external disturbance—thereby increasing response diversity. These landscape components or green spaces should be a variety of different habitat types (e.g., a variety of ecosystems), in different growth stages of each habitat type—relating to the four phases of the adaptive cycle, in different ownership types, and in different composition and spatial configuration. In landscape planning for conservation of biodiversity, the concept of functional diversity can translate to, for example, replicating connectivity across and within scales (e.g., Li et al. 2005a). For example, several streets with bioswales (i.e., vegetated swales) at the neighborhood scale can constitute a green corridor at the city scale, which, in turn, with city parks, can be a part of a regional park system at the urban regional scale (Jim and Chen 2003).

The focus of the research is the spatial configuration of land use, especially, green spaces, in urban regions—how green spaces can be best configured to increase the resilience of an urban region by applying ecological concepts such as response/functional diversity, redundancy, and connectivity across scales. The planning concepts and strategies described in chapter 4 are ways to deal with inevitable surprises and disturbances and to increase the resilience of a landscape. Variability in the landscape is the key to maintaining renewal capacity when the landscape undergoes some change (Holling et al. 2002b). Planning and design should develop landscapes that are (1) more spatially heterogeneous—not fragmented but integrated at multiple scales—by adding fine-scaled elements such as hedgerows, wind breaks, pockets of restored nature, diverse crops, integration and preservation of cultural/historical heritage (stone walls, monuments), etc. and (2) more functionally diverse (i.e., multifunctional landscapes) across and within scales. By so doing, response and functional diversity will be increased, making a landscape less sensitive to disturbances and building its capacity for enhanced resilience (Holling and Gunderson 2002).

5.5 Discussion

5.5.1 Urban Regions as Complex Adaptive Systems

Cities, urban regions, ecosystems, and landscapes arguably need to be seen as complex adaptive systems (Lansing 2003, Levin 2003, Waltner-Toews et al. 2003, Norberg and Cumming 2008), with attributes such as self-organization, adaptation, multiple spatial and temporal dynamics, to which systems theory (Lansing 2003, Levin 2003, Norberg and Cumming 2008) and resilience thinking (Holling 1973, Gunderson and Holling 2002, Gunderson and Pritchard 2002, Berkes et al. 2003, Pickett et al. 2004, Walker and Salt 2006, Woodward 2008) are applicable for developing more sustainable environments. Complex, multidirectional, continuous and dynamic processes (e.g., land development, land abandonment, forest loss, forest regeneration, population increase or decrease, water and species movement) operate in an urban region, which exists in cross-scale interactions. To manage and plan for the resilience of an urban region, at least three nested scales are recognized: neighborhood, city, and urban regional scales. An urban region, in turn, is nested in even broader biogeographical and socioeconomic areas that are known as megaregions (America 2050). Landscape planning at the urban regional scale would require collaboration among local and county governments, other stakeholders such as transportation planning authorities and land trust, a clearly agreed regional vision and goals, an entity specialized in regional planning issues with political "teeth" and budget (e.g., the Metro in Portland, OR), and a strong leadership from the regional entity (Wheeler 2000, Ndubisi 2008).

5.5.2 Biodiversity Hierarchy and Urban Regions as a Unit of Planning

Landscape diversity is the highest level in the biodiversity hierarchy according to Noss (1990). As landscapes are composed of ecosystems, which in turn are composed of species, which in turn are composed of genes, planning decisions made at the landscape level (or, the urban regional scale) have implications for the lower levels in the biodiversity hierarchy. According to the hierarchy theory (Allen and Starr 1982, O'Neill et al. 1986, Allen and Hoekstra 1992, Levin 1992, King 1997), one level of a hierarchy is governed by the higher level; the level below provides mechanisms that explain the higher level. When merging the nested hierarchy in biodiversity and the hierarchy theory, the species-level diversity (in chapter 3, forest birds were an example of species-level diversity) has implications for community/ecosystem diversity and genes diversity. (The distinction of a *nested* hierarchy as opposed to a hierarchy is made in section 4.1.2.2.)

I have found that the urban region is the relevant scale of planning/design/management, especially for species such as forest birds that have a large home range and a long dispersal distance. The ecological/spatial analysis in chapter 3 has, therefore, significance for planning the conservation of these species at the urban regional scale. The urban regional or landscape scale is an important scale for biodiversity conservation because it is at this scale that human influences on natural systems, and resulting landscape structure changes (including the spatial configuration of land uses) have consequences for ecosystem functions (Grimm et al. 2008). Land-use plans at the urban regional scale are arguably necessary to develop “smartly” (Benedict and McMahon 2002, Waddell 2002, Randolph 2004), lessening the impact of land use on biodiversity, mitigating the loss, and even creating new habitat. Therefore, land-use plans for urban regions arguably need to explicitly integrate the conservation of biodiversity as a recognized priority (Washitani and Yahara 1996, Ahern et al. 2006). A regional planning vision can then be used to coordinate planning efforts at the lower administrative levels.

A recent example of conservation planning at the regional scale, or even at a larger megaregional scale is seen in the Doris Duke Charitable Foundation’s commitment, in late October, 2009, of a grant of $400,000 over three years to the Regional Plan Association for wildlife conservation in the Northeast Megaregion in the U.S. as part of the America 2050 initiative (Doris Duke Charitable Foundation and Regional Plan Association 2009). The funds will be applied to a new project to improve the integration of nature conservation with land use planning and infrastructure investments in 13 states across the Northeast, from Maine to Virginia. This marks the first effort to coordinate regional landscape conservation at the megaregion scale, mirroring similar large-scale efforts focused on transportation planning and advocacy that are underway in the Northeast Megaregion (Doris Duke Charitable Foundation and Regional Plan Association 2009).

5.5.3 Meta-model Applications

The meta-model was developed as a tool for urban and regional planners to address various planning goals and issues in complex, adaptive social-ecological systems such as urban regions. It has its conceptual basis on the panarchy concept (Gunderson and Holling 2002) as a way to address the four phases of an adaptive cycle and interactions across scales (section 4.1.1.4.3). The key to the panarchy (i.e., linked hierarchically nested adaptive cycles connected by multi-scale dynamics) is dynamic spatial and temporal processes that connect multiple scales. The proposed meta-model focuses on these cross-scale processes, whether ecological processes such as dispersal and pollination, or social processes such as neighborhood tree planting programs gaining momentum and spreading to higher scales, or social memory.

The proposed meta-model can be applied to other planning units such as urban watersheds and sub-watersheds and planning goals such as water resource conservation and management. Urban watersheds and sub-watersheds share similar dynamics as urban regions with multiple actors and organizations, and dynamic processes operating at multiple spatial and temporal scales. Bryant (2006) discusses biodiversity conservation in the Cameron Run watershed in the context of greenway efforts at local and metropolitan scales. The analysis of the watershed found a potential to develop greenways using forested riparian corridors with patches having the potential for interior habitat. Stakeholder meetings identified various demands on the watershed and potential greenway development such as recreation, environmental education, aesthetics, and opportunities for biodiversity conservation. Moreover, urban watersheds such as the Cameron Run watershed are an important part of a regional natural areas network (Bryant

2006). Bryant (2006) argues that ecological greenways can be used to support urban biodiversity conservation.

Similar to the percentage of forest cover in an urban region, I argue that the percentage of impervious surfaces in an urban watershed is a slow-changing variable with thresholds, which affects both physical and biological measures of stream quality (Schueler 1994, Booth and Jackson 1997, May et al. 1997, Wang et al. 2001, Miltner et al. 2004). Thus, the application of the meta-model to urban watershed planning would identify the percent imperviousness as a key variable, and at various spatial scales such as sub-watersheds and reach segments it should be monitored in the proposed landscape planning best practice model (Figure 4.5) to help develop scenarios and to be used in proactive planning. Planning scenarios are used to provide alternative futures of an area (Steinitz et al. 2003) and address “what if” questions: what if the current level of land consumption is continued for the next 20 years? What if smart growth initiatives are adopted and future development is strategically clustered with infill development and open space is conserved? The percentage of impervious surfaces in an urban watershed can be associated with each scenario such as 25% for the unrestricted growth and 8% for the smart growth scenario. These figures can give the public and policy-makers the images of likely futures associated with each scenario based on the empirical evidence of the impervious surfaces’ impact on stream conditions. Since 10-15% seems to be the percent impervious surface threshold over which stream quality starts to deteriorate (Schueler 1994, Booth and Jackson 1997, May et al. 1997, Wang et al. 2001, Miltner et al. 2004), the threshold value can be used as a warning sign, when monitored, before negative effects are observed. The threshold value can also serve as a useful target for reducing the average imperviousness in the watershed below the threshold level. Proactive planning involves anticipating possible outcomes, and, when they are not desirable, acting to prevent them before they become the reality; therefore, it involves taking actions before the amount of imperviousness in a watershed exceeds the threshold level, or the threshold value can serve as a useful target for retrofitting. When new development is expected, planners can collaborate with developers, the city officials, and scientists, with the support of the other stakeholders, to have small "experiments" with varying percentages of impervious surfaces in various spatial configurations and monitor their effect on stream conditions. The monitoring results can form a scientific basis to develop land-use planning policies based on the certain percentage and spatial configuration of impervious surfaces. Then, the results should help formulate and test other percentages and/or spatial configurations to further reduce the negative impact in an adaptive "learning by doing" context (Kato and Ahern 2008).

5.5.4 Urban Regional Planning for Sustainability

It is at the regional scale where top-down planning (e.g., national planning) and bottom-up planning (e.g., community and neighborhood planning) meet. I argue that to develop effective regional visions and plans for growth and conservation, both top-down and bottom-up planning are necessary, and regional planning is the key. Forman (2008) captures the importance of regional planning in the context of globalization and the importance of local organizations and individual actions by the phrase: "Think globally, *plan regionally*, and act locally." Forman (2008) argues that urban regions are an appropriate planning and design unit to consider the implications of complex interactions between natural and human systems. Similarly, Musacchio (2009) argues that "the region is the appropriate scale for the study and implementation" of the six tenets (i.e., environment, economic, equity, aesthetics, experience, and ethics) of landscape sustainability. The global scale is too broad a scale to tackle the issue of sustainability, which requires international collaboration and cooperation (although by no means lessening the necessity to deal with this issue at this scale); the local scale is too fine a scale to capture the connections among patterns and processes (Musacchio 2009). There is a growing consensus among researchers that region should be the primary target of the study and planning of complex human and natural systems (Forman 2008, Grimm et al. 2008, Ndubisi 2008, Wu 2008, Musacchio 2009). Region is an appropriate unit for planning issues that span jurisdictional boundaries and/or for issues which collaborative planning efforts are necessary for successful achievement of their planning objectives. These issues appropriately addressed at the regional scale include: environmental and biodiversity conservation, affordable housing, urban growth boundaries, water resource planning and management, infrastructure, transportation, and sustainability (Wheeler 2000, Forman 2008).

My research has addressed biodiversity and green spaces in the context of ecological sustainability (Termorshuizen et al. 2007). Although its focus is on the abiotic and biotic components of the environment, it needs to integrate human activities and institutions (i.e., social and economic aspects) with the environment to address the issue of sustainability in a comprehensive manner (Beatley 2009). My research should be positioned relative to other fields of planning such as transportation planning, economic development planning, community development planning, and social, equity planning because addressing these areas of planning simultaneously in a proper context helps us develop sustainable landscapes. Urban regions are arguably the medium and the "battleground" for expressing cultural values, ethics, and testing innovative planning and design concepts based on landscape ecology, and should be the primary target of the study and planning of complex human and natural systems.

5.6 Synthesis and Conclusion

I have conceived this research from the perspective of landscape planners working with a transdisciplinary team of landscape ecologists, natural resource managers, policy-makers, other stakeholders and citizens. I am interested in the application of landscape ecology theories and principles to landscape planning (i.e., landscape ecological planning). So, the question is: Given the available ecological data, how can landscape planners develop plans/projects that achieve goals such as conservation of biodiversity and other related functions that the given spatial configuration of land uses can accommodate? I argue that there are opportunities here for: (1) interdisciplinary collaboration among related academic disciplines and professions such as landscape/land-use planners and landscape ecologists, planners and designers, and social scientists and ecologists towards the common goal of developing sustainable landscapes; (2) transdisciplinary approach to planning (Tress et al. 2003, 2005) to explicitly include the decision-makers, stakeholders, and citizens throughout the planning process; (3) devising "smart" land-use plans which address important social and ecological dimensions of sustainability; and (4) testing ecological hypotheses in an adaptive planning framework, with monitoring results being fed back to adapt the existing planning designs and even to adapt or reformulate goals and objectives.

The concept of multifunctionality is the key to addressing the question: what "collateral" functions and benefits (e.g., recreation, water quality, and cultural landscape protection) can be reasonably associated with the given amount and spatial configuration of forest cover? A multifunctional landscape allows the same planned space to serve multiple functions. For example, the Emerald Necklace in Massachusetts, U.S., was developed to serve multiple purposes (e.g., recreation, preservation of the natural landscape, and management of water quality) (Ndubisi 2002a, Fábos 1985, Fábos and Ahern 1996, Ahern 2004). In the land-use planning context, multifunctionality means that one landscape element at a certain spatial and temporal scale, for example, a suburban subdivision with 50 housing units, can be planned and designed to provide multiple functions such as human habitation, open space conservation, on-site water retention and infiltration, aesthetics, carbon sequestration, and native plant and animal habitat, by a certain spatial arrangement of housings, placement of infrastructure, porous pavement, integrated street open swales, etc. The subdivision, in turn, can be integrated into a regional development by light rails, for example.

Although the results from the spatial/ecological data analysis (chapter 3) cannot be extrapolated beyond the spatial extent(s), the selected forest bird species, and the amount and spatial configuration of forest within the study area, the selected forest breeding bird species can act as an indicator of forest interior conditions and the functioning of a healthy forest ecosystem, representing other associated species and functions/services (e.g., water retention and purification, evapotranspiration, temperature remediation, air purification, recreation, aesthetics, etc.) that a healthy forest ecosystem can provide. Some of these functions/services can arguably coexist with proper planning and management (Kato and Ahern 2009, section 2.4.6.3). I further argue that planning and designing for multifunctional landscapes, or allowing a landscape to function in a variety of ways is key challenge for creating sustainable landscapes in growing urban regions.

Given the specific amount and spatial configuration of forest cover, what collateral functions and benefits (e.g., recreation, water quality, and cultural landscape protection) can be reasonably associated with them then depends on the local and regional context, institutional settings, and abiotic, biotic, and cultural restrictions. Various planning and design strategies and concepts (Kato and Ahern 2009, section 2.4.6.3) can help alleviate the limitations and achieve desired functions. For example, integrating fine-grained landscape elements that add functions to the primary function such as crop production is one way to achieve multifunctional landscapes. Spatial arrangement such as connectivity helps provide multiple functions. For example, connected green-street corridors make possible functions such as jogging/bicycling, animal movement and dispersal, and water movement. The above example of different spatial arrangement of housing units in a subdivision allows open spaces to be clustered, which can be used for recreation, aesthetics, and water retention and infiltration.

5.6.1 Planning and Management Recommendations

Finally, I will discuss specific planning and management recommendations to plan and manage green spaces in urban regions, based primarily on chapters 3 and 4.

First, the important landscape metrics identified in the reduced models should be monitored as a part of the landscape planning best practice model. The landscape metrics identified as important for the conservation of the target bird species as a group are the percentage of forest cover in the landscape, contrast-weighted forest edge density, and similarity of land cover types to forest cover. The percentage of forest cover in an urban region directly relates to the urban tree canopy goal and vision. Forest edges and/or edge contrast can be intentionally created, reduced, and mitigated by land-use planning. The landscape matrix can be managed by paying attention to the similarity between land cover types (in terms of % canopy) and the aggregation of those patches of similar land cover types. These landscape metrics should be monitored as a part of the landscape planning best practice model so that they can be used (1) to assess the current state of the regional forest cover and (2) to detect land cover changes over time, acting as warning signs before thresholds are crossed and (3) to compare with other regions where the target species are more successfully managed to guide the overall planning effort.

Second, for those species that are more sensitive to forest loss and fragmentation, such as Ovenbird (OVEN) and Black-and-white Warbler (BWWA), to increase the number of individuals, specific open space planning strategies such as Colding's (2007) ecological land-use complementation (ELC) can work in concert with important landscape metrics such as contrast-weighted edge density (-) and similarity (+) to achieve better spatial configuration of green spaces. As the signs in the parentheses indicate, the abundance of OVEN and BWWA increases as forest edge density decreases and/or edge contrast decreases and as similarity of the land cover types in relation to forest cover increases. This indicates a landscape composed of land cover types similar to forest cover such as shrub, and these patches are more aggregated and their adjacencies include similar land cover types in terms of percent tree canopy. The ELC strategy can develop such a landscape by clustering different types of green spaces such as a golf course, a remnant forest patch, and an orchard to increase the total habitat area as well as to provide complementation/supplementation effects for the various needs of target species during its different life stages or during different activities such as foraging, nesting, and rearing the young. The ELC is developed for urban areas but I argue that the concept can be applicable to broader urban regions. When different land uses are considered at a coarse scale, such as at the urban regional scale, different green-space types are more similar to each other than other land cover types such as urban, industrial, and herbaceous. Then, at the urban regional scale, we can see the benefits of clustering more similar land cover types, in terms of the average tree canopy percentage, for example, to lessen forest edge density and/or edge contrast and increase functional connectivity.

Third, a multi-scale approach (e.g., the meta-model) is needed to achieve biodiversity conservation in urban regions because biodiversity encompasses multiple levels of biological organization from genes, populations/species, communities/ecosystems, to regional landscapes (Noss 1990). Because each species operates at a specific scale (Wiens 1989), conserving multiple species together requires a multi-scale approach (Wiens et al. 2002). Biodiversity concerns the whole, each level of the organization, and their inter-relations, and the actions at one level affects the levels both above and below. Therefore, the conservation of biodiversity calls for a multi-scale approach. The multi-scale approach is needed also because in planning and design, the hierarchy theory (Allen and Starr 1982, O'Neill et al. 1986) indicates that it is necessary to consider at least three scales to understand the larger context and details (mechanisms) affecting a plan and project at the target scale (Dines and Brown 2001). Also, because most disturbances operate within a specific rage of scales, to maintain and enhance the resilience of a system, planning strategies such as protecting landscape elements that support the same function across scales (i.e., functional diversity) would maintain the function even when the landscape elements at one scale are modified or destroyed by the disturbance (Gunderson et al. 2002a).

The multi-scale approach may be applied to other areas and scales (e.g., city and neighborhood) of planning. For example, urban watershed planning requires the multi-scale approach to address scale-specific issues at the sub-watershed and the reach scales. At the broader watershed scale, the percentage of impervious surfaces and the spatial configuration of land uses may affect the biophysical and geomorphological characteristics of the river (Frothingham et al. 2002). At the sub-watershed scale, different factors such as specific polluting sources and the coarse woody debris may be more important for the same characteristics. The multi-scale approach may be applied to transportation planning as well. Again, different issues and factors may be more important at different scales of transportation planning, for example, laying out the interstate highways, planning major state road networks, local traffic studies, and planning for emergency routes. Infrastructure such as roads and utility lines is by nature connected in a hierarchy. Planning for transportation networks requires thinking on multiple levels and the interactions between the levels over multiple temporal scales (Fineman et al. 2003).

Fourth, generally speaking, the planning concepts and strategies discussed in the dissertation and particularly in chapter 4 are a way to deal with inevitable surprises and disturbances and to increase the resilience of a landscape. Diversity and variability in the landscape is the key to maintaining renewal, or resilience capacity when the landscape undergoes some change (Holling et al. 2002b). This reorganization and renewal phase of the adaptive cycle of the landscape is critical for it to remain in the desired state (from the viewpoint of humans). However, human activities, in many parts of the world, have reduced diversity (including biodiversity) and variability of the system, so that a smaller amount of disturbances now can flip the system to an undesirable state. To increase variety in the landscape, planning and design should develop landscapes that are more spatially heterogeneous—not fragmented but integrated at multiple scales—by adding fine-scaled elements such as green infrastructure, hedgerows, wind breaks, pockets of restored nature, diverse crops, and integration and preservation of cultural/historical heritage. Incorporating the concept of adaptive cycles is one way to achieve the integration of heterogeneity at multiple, spatial and temporal scales (Gunderson and Holling 2002, Walker et al. 2004, Walker and Salt 2006). For example, urban regional forest cover can be managed at various stages of growth, with various sizes and spatial configurations. Planning concepts and strategies such as cross-scale connectivity, the Casco concept, and ecological networks confer the benefits of the integration of multiple scales (Ahern 1991, 1999, van Buuren and Kerkstra 1993, Jim and Chen 2003, Li et al. 2005a, von Haaren and Reich 2006, Jones-Walters 2007). For example, the ecological network concept allows its constituent elements (e.g., patches, stepping-stone habitats, corridors) to change (e.g., developed and/or spatially change the locations) while maintaining the goal of providing a conservation network for species (Opdam et al. 2006). This allows for the possibility of local population extinction and/or degradation of patches due to disturbance and development. To increase variety in the landscape, planning and design should also develop landscapes that are more functionally diverse (i.e., multifunctional landscapes) across and within scales. This would increase response and functional diversity, making a landscape less sensitive to disturbances and building its capacity for enhanced resilience (Holling and Gunderson 2002).

5.6.2 Future Research Directions

The effect of forest fragmentation per se can be investigated by separating the data above and below the strongly identified thresholds, accounting for the effect of forest amount, and running the multiple regression analysis to see if the connectivity (or isolation) indices will be identified as important in the reduced models, especially below the thresholds. For example, for BWWA the data can be split above and below 86% forest cover at the 180 m and 6 km buffer sizes. Since the two-breakpoint models have found another threshold at around 23% forest cover for BWWA at these scales, the data may have to be split in three segments and focus on the data below 23% forest cover to see if the connectivity metrics are chosen as important.

The logical next step of the research is to apply the proposed greenspace conservation planning framework to real urban regions such as the Greater Boston region and the Portland metropolitan region. The new round of urban long-term ecological research areas (ULTRA) may offer specific opportunities for testing and adapting this model. The application would allow me to test and further develop hypotheses for different spatial configurations of green spaces to develop more smartly, and their effects on ecological processes. Teaming up with developers, city officials, and ecologists to test various spatial configurations of green spaces in an adaptive manner (at the next development of similar subdivisions or retrofitting projects) would be an excellent example of interdisciplinary collaboration and practice with the adaptive component. The research results should be fed back to the next development with transdisciplinary practice. I am also interested in testing the meta-model in an international context—the concepts and principles should be applicable to any major urban region around the world. Applying the meta-model to these urban regions to develop cases is the next logical step in my research and I believe I am well positioned to make significant research resulting in publications.

The next 50 to 100 years will see unprecedented landscape changes with rapidly increasing global population, rapid loss of biodiversity, peak oil, climate change, widening income gap between the rich and the poor, and between rich and poor countries (Homer-Dixon 2006, Friedman 2008). The list of concerns goes on and on. It will be a time of great challenge for planners as they struggle to achieve more sustainable landscapes and regions. At the same time, these issues provide many opportunities in which planners can contribute to make a positive difference. For example, opportunities and challenges to conserve biodiversity lie in the urban regions in the U.S. where the new population growth of 100 million is expected occur in the next 30 years (Nelson and Lang 2007)—to create an environment where humans and nature can mesh together, live in a long term (Forman 2008). I believe the proposed meta-model, developed based on the concepts of panarchy and resilience, will serve urban and regional planners as a useful tool for them to recognize cross-scale ecological and social processes, and planning concepts and strategies that they can use to achieve social equity, environmental, and economic planning goals.

APPENDICES

APPENDIX A

GLOSSARY

Abiotic

Nonliving; the physical and chemical components of an environment that result in particular distributions and abundances of organisms (adopted from Spray and McGlothlin 2003).

Biodiversity (Biological Diversity)

1: Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (adopted from the Convention on Biological Diversity, Section I, Article 2, 1992).

2: Broadly defined, the diversity of life at all levels of organization from the gene to the landscape, and all the interconnections that support life. More pragmatically, species and communities found in their natural places, distributed and functioning within their natural range of variability (adopted from Leitão et al. 2006, p. 211).

3: Biodiversity is the totality, over time, of genes, species, and ecosystems in an ecosystem or region, including the ecosystem structure and function that supports and sustains life (adopted from Ahern et al. 2006, p. 6).

4: Biodiversity encompasses multiple levels of biological organization (Noss 1990, Peck 1998, Dale 2001, Groom et al. 2006). Noss (1990) expanded on the three primary attributes of biodiversity recognized by Franklin et al. (1981)—composition, structure, and function—into a nested hierarchy that incorporates elements of each attribute at four levels of organization: regional landscape, ecosystem-community, species-population, and genetic. Noss (1990) proposed that measurable indicators of each attribute at the four levels of organization be selected and monitored to assess the status of biodiversity over time.

Biological Community

A biological community, or community, is all the species that occupies a particular locality and the interactions among those species (adopted from Primack 2008, p. 26).

Biotic

Living; those living components of the environment that result in particular distributions and abundances of organisms (such as competition for food, space, and mates) (adopted from Spray and McGlothlin 2003, p. 164).

Class Level

Pertaining to a single patch type (land cover type) in a categorically classified landscape or, as in a hierarchy, the aggregation of patches of the same type into classes (adopted from Leitão et al. 2006, p. 211). Class-level metrics (FRAGSTATS term) quantify characteristics of an entire class (i.e., patch type), such as total extent, average patch size and degree of aggregation or clumping, and return a unique value for each class (i.e., one record per class) (adopted from Leitão et al. 2006, p. 21).

Connectivity

The spatial continuity of a patch type (or class) across a landscape (i.e., structural connectivity) or the degree to which specific ecological flows (e.g., movement of energy, materials, and organisms) across a landscape are facilitated or impeded (i.e., functional connectivity) (adopted from Leitão et al. 2006, p. 211).

Conservation Planning

Conservation planning is a branch of landscape planning whose primary goal is to conserve biodiversity at the appropriate scale (spatial and temporal) for the target biodiversity. In short, it is planning for biodiversity conservation (Margules 1999). Land use planning and conservation planning should be practiced together (Walmsley 2006), for both have goals of balancing conservation and development. Conservation planning historically focused on the design of reserve networks (Noss and Daly 2006) and designating protected areas but is lately shifting its focus to the planning and management of the landscape matrix as well, surrounding the protected areas (Margules and Pressey 2000, Sarkar et al. 2006), and of ecosystems, embracing uncertainty and change and applying adaptive management and planning and the concept of resilience (Lister and Kay 2000). “Conservation planning must become a more flexible, resilient, and adaptive process, based on proactive, collaborative learning and rooted in an interdisciplinary (and perhaps even transdisciplinary) art and science” (Lister and Kay 2000, p. 211).

Disturbance

Since some disturbances are part of natural disturbance regime such as the fire disturbance regime, in this dissertation, I adopt White and Pickett’s (1985) more value-neutral definition of disturbance which includes “environmental fluctuations and destructive events, whether or not these are perceived as “normal” for a particular system” (p. 6). From the perspective of biological systems, they argue that disturbance is relative to the spatial and temporal dimensions of the system at hand—for example, relative to the size and the lifespan of the dominant organisms of the biological community of interest.

Ecological Function

See ecological process.

Ecological Network

Ecological networks can be defined as systems of nature reserves and their interconnections that make a fragmented natural system coherent, so as to support more biological diversity than in its non-connected form. An ecological network is composed of core areas, (usually protected by) buffer zones and (connected through) ecological corridors (adopted from Jongman 2004, p. 24).

Ecological Planning

1: Steiner (2000) defines ecological planning (or applied human ecology) as "the use of biophysical and sociocultural information to suggest opportunities and constraints for decision making about the use of the landscape" (pp. 9, 10).

2: Ecological planning includes diverse activities from "the development of algorisms to optimize the design of conservation reserve networks that will maximize native biological diversity, to designs for housing developments that reduce urban sprawl by creating compact neighborhoods with protected open space, to regional plans that project alternative future scenarios of land use change" (Collinge 2009, p. 246). These diverse activities have a common denominator: "the integration of ecological knowledge with intentional human actions to direct spatial patterns of environmental change"

(Collinge 2009, p. 246). In essence, Collinge's (2009) concept of ecological planning includes all the activities under landscape ecological planning (see below).

Ecological Process

Throughout this dissertation, ecological processes and functions are used interchangeably to mean broadly the flow of water, energy, materials, and organisms; the interactions among organisms such as predation, symbiosis, and mutualism; and the interactions between organisms and the environment (Forman and Godron 1986, Forman 1995, Benedict and McMahon 2006). Other examples of ecological (ecosystem) processes or functions include: pollination, seed dispersal, the decomposition of dead organic matter, carbon sequestration, and water filtration (Collinge 2009).

For the same token, landscape processes are used interchangeably with landscape functions. Landscape processes and ecological processes are used to mean basically the same phenomena; ecological processes may sometimes be used to restrict the phenomena involving organisms. Also, as below, ecological processes and ecosystem processes basically mean the same.

Ecosystem

An ecosystem is a community of living organisms together with the physical processes that occur within an environment (adopted from Pullin 2002). There are abiotic (non-living) and biotic (living) components of an ecosystem, all potentially interacting to form a functioning unit, distinguishable, although not isolated, from other ecosystems (Pullin 2002, Spray and McGlothlin 2003). An ecosystem is a biological community together with its associated physical and chemical environment (adopted from Primack 2008, p. 301).

Ecosystem Function

Ecosystem function is a general term referring to the suite of processes, such as primary production, ecosystem respiration, biogeochemical transformations, information transfer, and material transport, that occur within ecosystems and link the structural components (adopted from Grimm et al. 2000).

Ecosystem Service

Ecosystem or ecological services are the benefits people obtain from ecosystem (ecological) processes (Daily 1997, MA 2005). These include water and air purification, flood control, erosion control, generation of fertile soils, detoxification of wastes, resistance to climate and other environmental changes, pollination, and aesthetic and cultural benefits that derive from nature (Andersson et al. 2007).

Edge Effects

Altered environmental and biological conditions at the edges of a fragmented habitat (adopted from Primack 2008, p. 301). Examples include greater fluctuations in levels of light, temperature, humidity, and wind (Laurance et al. 2002).

Environmental Planning

Environmental planning, as a sub-filed of regional planning, includes all the planning and management activities where the emphasis is on environmental considerations (e.g., clean air and water) rather than other factors (e.g., social, cultural, or political) (Forman 1995, Marsh 2005). Following the environmental crisis of the 1960's and 1970's in the U.S., the environmental movement used the term "environment" to mean "things of natural origin in the landscape, that is, air, water, forests, animals, river valleys, mountains, canyons, and the like" (Marsh 2005, p. 3). Environmental planning applies to both environmental protection (e.g., protection of natural resources) and solving environmental problems (e.g., air and water pollution) (Randolph 2004).

Focal Species

Plant and animal species that are critical to maintaining ecologically healthy conditions (adopted from Benedict and McMahon 2006, p. 281); species whose requirements for persistence include attributes that must be present for a landscape to meet the needs of most of the species in the given area (adopted from Ahern et al. 2006, p. 15). Therefore, focal species are often used to determine maximum acceptable levels of threat (Ahern et al. 2006). This is an extension of the umbrella species concept (Ahern et al. 2006).

Fragmentation

Landscape process in which a patch type (e.g., habitat type or land cover type) is progressively subdivided into smaller, geometrically altered, and more isolated fragments, often as a result of both natural and human activities (adopted from Leitão et al. 2006, p. 212). Fragmentation per se refers specifically to the breaking up of a patch type into smaller, disconnected fragments, and is (should be) distinct from the loss of patch area per se, which may or may not occur concomitantly with fragmentation (adopted from McGarigal and McComb 1999).

Greenway

Greenways are networks of connected linear open spaces along natural or human-made features such as rivers, ridgelines, railroads, canals or roads. They are planned, designed and managed to connect and protect ecological, scenic, recreational and cultural/historic resources. A greenway can serve for multiple purposes that are compatible with the concept of sustainable land use. (Little 1990, Ahern 1995, Erickson 2004).

Greenway goals can be broadly categorized into two objectives: to provide ecological and social/cultural functions (Fábos and Ahern 1996, Erickson 2004). Ecological functions include: protection of water quality (Erickson 2004), protection of natural and environmentally fragile areas (Erickson 2004, Bryant 2006, Imam 2006, von Haaren and Reich 2006), mitigation/lessening of the effects of habitat fragmentation on wildlife (Smith 1993a, Flink and Searns 1993, Bryant 2006, von Haaren and Reich 2006), and conservation of biodiversity, which is arguably the most important greenway goal in terms of sustainable landscape planning (Ahern 1995). The effectiveness at which greenways support these goals varies according to their width, shape, location, context, and other factors (Smith 1993b). Social/cultural functions of greenways include: recreation, transportation, aesthetic enhancement, protection of significant historical and cultural sites, and environmental education (Smith 1993a, Erickson 2004, Imam 2006, Ribeiro and Barao 2006).

Green Infrastructure

1: Also called ecological infrastructure, “our world’s natural life-support system—an interconnected network of waterways, wetlands, woodlands, wildlife habitats, and other natural areas; greenways, parks, and other conservation lands; working farms, ranches, and forests; and wilderness and other open spaces that support native species, maintain natural ecological processes, sustain air and water resources, and contribute to the health and quality of life for communities and people” (adopted from Benedict and McMahon 2006, pp. 281, 282).

2: Green infrastructure is an emerging planning and design concept that is principally structured by a hybrid hydrological/drainage network, complementing and linking relict green areas with built infrastructure that provide ecological functions (adopted from Ahern and Kato 2007, p. 287). It also includes designed elements to treat storm water and enhance biodiversity such as rain gardens, bioswales, rooftop gardens, etc.

Greenway Planning

A subset of landscape planning, focused on the elements that constitute greenways, including: large protected areas, riparian corridors, other corridors, and linkages. Greenway planning is usually imbedded within a comprehensive planning approach which addresses the other concerns/sectors of planning, including: physical, economic, and social (adopted from Ahern 2002). While the definitions of greenways and their primary purposes vary among different counties and specific areas to which greenways are applied, the most succinct definition of greenway planning is "planning linear corridors of protected green space at multiple-scales and for multiple purposes" (Fábos and Ryan 2006).

Habitat

Habitat refers to "the place where an animal or plant normally lives, often characterized by a dominant plant form or physical characteristic (that is, the stream habitat, the forest habitat)" (Ricklefs and Miller 2000, p. 731). Habitat therefore includes the necessary resources and conditions for specific organisms for their specific purposes such as foraging and nesting (Ricklefs and Miller 2000).

Institution

Institutions are the formal structures that codify patterns of human behavior (Grimm et al. 2000).

Interdisciplinary

An interdisciplinary project is one that involves “several unrelated academic disciplines in a way that forces them to cross subject boundaries to create new knowledge and theory and solve a common research goal” (Tress et al. 2005, p. 17). Unrelated disciplines have contrasting research paradigms such as qualitative and quantitative approaches, or analytical and interpretative approaches (Tress et al. 2005). An interdisciplinary approach, therefore, means reaching out beyond one discipline’s “turf” and really reaching out to other academic disciplines to the extent that its content and boundary is redefined.

Intrinsic Suitability

The inherent capability of an area to support a particular land use with the least detriment to the economy and the environment (adopted from Steiner 2000, p. 428).

Landscape

When the term is used in landscape ecological studies, a landscape can be defined as a heterogeneous land area composed of interacting ecosystems that repeat in a similar form under similar climate, geomorphology, and disturbance regimes (Forman and Godron 1986, Forman 1995, Turner et al. 2001). Leitão et al. (2006) note that the landscape concept differs from the traditional ecosystem concept in that it focuses on groups of ecosystems and the interactions among them. When the term is used in a planning context, I adopt Steiner’s (2000) definition: “The composite features of one part of the surface of the earth that distinguish it from another area…..The landscape encompasses the uses of land—housing, transportation, agriculture, recreation, and natural areas—and is a composite of those uses. A landscape is more than a picturesque view; it is the sum of the parts that can be seen, the layers and intersections of time and culture that comprise a place—a natural and cultural palimpsest" (p. 4). The definition by the European Landscape Convention (2000) is an example that has a wider perspective: "Landscape means an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors." In my view, Tress and Tress (2001) provide the most evolved landscape concept. The transdisciplinary landscape concept is built on the five dimensions: (1) spatial, (2) mental, (3) temporal, (4) a nexus of nature and culture, and (5) a complex system. They view landscape as being composed of coexisting subsystems: the geo-, bio-, and noo-sphere. The three spheres are interrelated, creating a complex, dynamic system. Holism provides a useful concept to study this complex whole. Its utility in landscape ecological planning study is its assumption: one can study the whole without necessarily knowing all the parts (Zonneveld 1990, Ndubisi 2002a). Tress and Tress (2001) present a conceptual model (i.e., the people-landscape interaction model) to describe the transdisciplinary landscape concept. A similar, conceptual model of human-environmental interactions in the landscape is proposed by Gobster et al. (2007). The characteristic of Tress and Tress' (2001) model is that people are both part of the landscape and relating themselves to landscape from the noosphere—the mental space where people can reflect on their actions. Tress and Tress (2001) acknowledge that they have developed their ideas based on Naveh (2001), who gave basically the same view of a landscape (i.e., the "Total Human Ecosystem"). The transdisciplinary landscape concept and the people-landscape interaction model provide a conceptual basis for studying complex interrelations that occur in a landscape and can serve as a framework for future landscape research (Tress and Tress 2001).

In conclusion, researchers seems to agree that a landscape is a complex adaptive system composed of interacting physical, biological, and cultural systems. The definitions from a landscape planning perspective see landscape as the product of age-long interactions between humans and nature. The interaction is ongoing and landscape keeps changing. Lately, humans are the major driving force of this landscape change by consuming landscapes for various land uses. Humans are in turn affected by the existing conditions of the landscape and also make mental associations (cognitive maps, memories, bonds, place attachment, etc.) with the landscape and preference based on aesthetics. Human perceptions of the landscape and ethics they carry play a large role in how the landscapes are developed and managed, together with ecosystems and species that are part (Wiens 2005).

Landscape Composition

Landscape composition refers to the variety and abundance of patch types without regard to their spatial character or arrangement (adopted from Leitão et al. 2006, p. 20).

Landscape Configuration

Landscape configuration refers to the spatial character and arrangement, position, or orientation of landscape elements (adopted from Leitão et al. 2006, pp. 20, 21).

Landscape Ecology

Landscape ecology is the study of the relationship between landscape structure (i.e., composition and configuration of landscape elements) and function (i.e., ecological processes) over a heterogeneous landscape at a broad spatial scale (Forman and Godron 1986, Turner 1989, 2005, Turner et al. 2001). Landscape ecology also examines how the relationship changes across spatial and temporal scales (Forman and Godron 1986, Wiens 1989). Spatial heterogeneity is the key (Pickett and Cadenasso 1995, Turner 2005) but temporal change is also important in urbanizing areas (e.g., faster rate of turnovers of land use/cover types).

Landscape Ecological Planning

1: Simply put, landscape ecological planning is a branch of landscape planning that has attempted to integrate landscape ecology theories and principles into landscape planning. Leitão et al. (2006) define landscape ecological planning as "planning for ecologically sustainable landscapes; considering the spatial structure of the system, the flows of energy and materials among system components and between the system and its surroundings, and the evolution of the system over time—explicitly including the values, actions and impacts of humans" (p. 245). Landscape ecological planning is a way to address sustainability with the landscape as the principle unit.

2: A contemporary approach to landscape planning, based specifically on theory and principles from landscape ecology. Landscape ecological planning integrates topological and chorological perspectives to achieve a dynamic understanding of landscape pattern-process relationships. Landscape ecological planning uses the patch-corridor-matrix model, from landscape ecology, and recognizes the inherent benefits of connectivity. Landscape ecological planning addresses the inherent uncertainty of site-specific ecological information through an adaptive approach in which monitoring and analysis are performed to determine if the planning action(s) achieved the intended results (adopted from Ahern 2002).

Landscape Planning

1: Landscape planning is defined as a resource allocation and planning activity, dealing with landscape features, processes, and systems, for the sustainable use of resources at a broad spatial scale (Cook and van Lier 1994, Ndubisi 1997, Ahern 1999, Marsh 2005). According to Marsh (2005), landscape planning is a subfield of environmental planning. While environmental planning primarily deals with "things of natural origin" (Marsh 2005, p. 3), human-landscape interactions are central in landscape planning (Cook and van Lier 1994, Forman 1995, Ndubisi 1997). Because a landscape, which is the object of planning, encompasses complex interactions between human activities and ecological processes, landscape planning has necessarily developed methods to deal with making priorities among multiple competing land uses and strategies to combine compatible uses based on the intrinsic suitability of the land for different land uses.

2: Landscape planning is a process of managing change while maintaining regard for the wise and sustained use of the landscape, based on the knowledge of the reciprocal relationships between people and land (adopted from Ndubisi 1997)—a focus on the reciprocal relationships between pattern and process, i.e., structure and function, and between the natural and social systems.

3: Landscape planning provides information about the existing qualities of the landscape (i.e., landscape potentials), their value and sensitivity to the existing and potential impacts on these potentials, and the objectives and guidelines for the development of the landscape, upon which proposed measures and development plans can be measured (adopted from Mander 2008, pp. 2116-2126).

Landscape Structure

Landscape structure encompasses the characteristics of landscape elements and is composed of landscape composition (i.e., what and how much)—not in terms of types/density of trees/canopy cover—and configuration (i.e., where). Although landscape "pattern" is often used by some researchers (e.g., Turner et al. 2001) to mean landscape structure, I have used landscape pattern to mean only landscape configuration (i.e., the spatial configuration of landscape elements such as ecosystems and land use/cover), not both composition and configuration. The only exception to this rule is where I have used the term pattern in the context of the pattern-and-process relationship (as used and popularized by Turner 1989), in which case, landscape pattern has the same meaning as landscape structure.

Land Use Planning

Land use planning is the systematic assessment of the intrinsic capability of the land and alternative land use and socio-economic conditions in order to select and adopt the best land use options (Food and Agriculture Organization [FAO] of the United Nations 1993, p. 96). These land uses should meet the current needs of the people without compromising those of the future (FAO 1993). Land use planning should seek to improve the current conditions of land and anticipate land use change (FAO 1993).

Metropolitan Statistical Area (MSA)

Metropolitan statistical areas are defined in two ways: a city of at least 50,000 population or an urbanized area of at least 50,000 population with a total metropolitan area population of at least 100,000. MSAs are defined in terms of whole counties, except in the six New England states where they are defined in terms of cities and towns. In addition to the county containing the main city, an MSA also includes additional counties having strong economic and social ties to the central county (adopted from Steiner 2000, p. 429).

Patch

A patch is a relatively homogeneous area that differs from its surroundings. In the dissertation, the term is used to mean a habitat patch for the species of interest such as forest birds.

Proactive Strategy

Proactive strategies mean that actions are taken before a problem arises; "conservation and assessment efforts undertaken before a problem arises or before a problem is beyond mitigation. An example is the National Gap Analysis Program." (Ahern et al. 2006, p. 97)

Response Diversity

Response diversity refers to the multitude of responses to environmental change and disturbances, among species contributing to the same ecosystem function. This kind of diversity plays a crucial role in sustaining the resilience of ecosystems to cope with disturbance and change. If all species within a functional group (e.g. pollinators, seed dispersers or decomposers) are equally sensitive to a particular disturbance, the system will have low response diversity and be vulnerable to that particular disturbance (adopted from Stockholm Resilience Centre 2007).

Sustainable Landscape Planning

Sustainable landscape planning strives to achieve a long-term (i.e., over decades or human generations) and productive balance between natural systems and the human use of these systems (Marsh 2005, Forman 2008).

Transdisciplinary

A transdisciplinary project is one that integrates both academic researchers from different unrelated disciplines and non-academic participants, such as land managers and the public, to reach a common goal and create new knowledge and theory (Tress et al. 2003, 2005); a planning process that encourages active public participation and the involvement of stakeholders in the planning process along with interdisciplinary collaboration and integration of various disciplines (e.g., architecture, landscape architecture, planning, civil engineering, ecology, sociology, economics, psychology, etc.). See Tress et al. 2005 for more discussion.

APPENDIX B

BBS AVEARGE ROUTE-LEVEL ESTIMATES

Table B: Average bird abundance for the selected routes over 2002-2006. The selected species were recorded in at least two of the five years. Rteno: Route number; EAWP: Eastern Wood-Pewee; WOTH: Wood Thrush; BWWA: Black-and-white Warbler; AMRE: American Redstart; and OVEN: Ovenbird.

Rteno	EAWP	WOTH	BWWA	AMRE	OVEN
44001	2.25	4.00	2.25	1.50	7.75
44002	3.33	1.67	3.00	2.33	12.00
44003	1.60	0.40	2.40	3.00	23.20
44004	1.00	0.60	0.80	2.60	3.80
44022	1.50	0.50	3.00	2.00	14.50
58001	5.40	4.40	3.00	2.20	10.20
58004	4.20	5.20	2.20	4.20	8.60
58006	3.00	6.40	5.40	5.20	21.40
58008	7.00	10.20	1.60	4.00	26.40
87022	0.80	2.60	0.20	0.20	4.80
47001	0.80	0.60	0.40	0.00	0.60
47006	1.50	3.25	0.25	0.00	2.50
47007	2.00	1.00	0.00	0.50	2.00
47009	5.00	1.50	0.75	0.25	5.75
47011	1.00	4.40	0.00	0.20	2.20
47014	2.60	8.80	2.40	5.00	11.20
47015	3.20	7.40	2.80	7.00	12.00
47016	8.33	5.33	2.33	4.33	14.00
47017	7.25	11.25	16.00	7.50	38.25
47018	1.60	15.60	4.20	12.80	24.40
47019	2.80	16.80	15.00	20.20	50.20
47021	2.40	9.40	2.60	19.20	21.60
47022	8.20	13.80	4.00	21.20	31.60
47112	4.00	7.50	1.00	1.00	8.25
47113	4.50	5.25	0.75	0.00	10.50
47900	1.50	3.75	2.00	2.75	35.25
77800	0.00	0.00	0.00	0.00	0.00
18001	6.00	24.00	2.00	0.50	12.50
18003	4.00	8.40	0.40	1.80	8.40
18006	1.50	8.00	0.00	0.00	3.50
18010	1.00	2.50	0.00	0.00	0.00
18011	1.00	5.00	0.00	1.00	2.50
18014	3.50	18.50	1.50	9.50	25.00
18015	1.00	5.50	0.00	0.00	0.50
61001	0.00	1.00	0.25	1.50	0.75
61002	0.00	0.50	0.00	0.00	3.50

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Table B, continued

61004	0.00	0.00	0.00	0.00	0.00
61005	0.33	0.33	0.00	0.00	0.00
61008	5.67	19.00	1.00	6.33	6.67
61009	5.40	11.60	1.00	5.80	0.80
61014	10.40	11.60	2.00	7.00	9.60
61017	1.50	9.50	1.00	8.50	19.50
61022	4.80	7.00	4.60	6.20	13.60
61025	4.75	9.00	2.00	2.00	6.75
61027	3.00	12.25	4.25	7.00	29.00
61038	0.60	3.40	0.20	0.20	5.60
61039	1.00	7.50	0.00	0.00	5.50
61043	2.00	5.67	0.00	1.30	5.00
61044	1.50	3.50	0.00	2.00	7.50
61050	8.33	15.00	0.00	0.00	6.67
61051	2.00	2.50	0.25	0.00	0.50
61053	1.50	1.75	0.00	0.75	0.00
61064	4.00	3.00	0.00	0.20	0.20
61065	2.60	5.00	0.00	2.40	3.00
61068	0.50	2.50	0.00	0.00	0.00
61070	0.50	5.50	0.50	4.50	4.00
61072	6.00	11.00	4.50	9.50	15.00
61073	0.75	0.75	0.50	9.50	31.75
61074	5.50	10.50	1.00	6.50	9.00
61075	0.80	9.80	0.40	6.80	22.20
61079	2.67	1.67	0.33	2.00	14.00
61080	2.75	6.75	0.00	2.25	1.25
61088	0.00	10.67	3.67	10.33	16.67
61089	7.60	17.00	1.80	3.60	7.60
61090	5.33	16.00	0.00	0.33	8.00
61091	3.67	8.67	0.00	3.00	1.67
61092	4.33	6.00	1.33	0.33	21.33
61093	3.40	7.00	2.20	2.40	33.00
61120	2.33	8.33	0.00	0.67	4.67
61207	1.40	6.80	1.40	1.40	4.40
59002	0.33	7.00	0.00	0.00	5.33
59003	2.40	4.00	0.00	0.00	7.80
59006	3.50	7.00	0.50	0.00	18.00
59010	1.80	5.80	0.40	0.20	2.20
59013	1.50	6.75	0.00	0.00	8.25
59014	1.50	6.50	0.00	2.00	3.50
59018	0.40	18.20	0.40	1.20	5.80
59019	12.20	13.20	1.20	14.40	21.20
59020	5.60	9.40	0.00	1.00	2.00
59021	15.00	18.40	2.20	16.40	20.20
59022	1.20	8.00	0.00	2.00	2.40
59023	4.00	12.00	0.00	0.50	9.00

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Table B, continued

59025	1.00	4.75	0.00	0.25	2.25
59026	0.00	3.00	0.00	0.00	1.33
59027	2.00	14.50	0.00	0.00	1.00
59029	3.40	6.60	0.20	0.00	11.60
59030	0.00	0.00	0.00	0.00	0.00
59104	2.40	12.60	0.00	0.00	3.20
59107	7.00	10.40	0.60	0.20	18.40
59116	10.50	12.50	1.00	4.25	11.00
59231	1.33	4.67	0.00	0.00	7.67
72046	1.50	10.00	0.00	0.00	0.50
72047	5.40	18.80	0.00	3.00	1.80
72048	4.00	10.00	0.00	0.00	0.00
72053	3.60	9.60	0.80	3.00	3.60
72056	6.20	22.40	1.20	4.40	14.20
72063	3.40	10.20	0.20	2.80	1.80
72076	1.40	8.60	0.00	0.20	0.60
72078	0.50	3.50	0.25	0.50	1.00
72084	3.20	19.80	0.00	0.00	0.20
72086	2.60	7.60	0.00	0.00	0.40
72097	1.00	11.20	0.00	0.00	1.40
72101	7.80	12.40	1.00	6.20	11.00
72126	2.60	1.00	8.00	12.40	18.40
72151	2.40	10.00	0.00	0.20	2.20
72165	17.40	22.60	1.20	7.80	38.00
72174	2.00	4.50	0.00	0.00	0.00
72181	0.50	11.50	0.00	0.00	2.00
72182	3.50	6.50	0.00	0.00	0.25
72189	11.67	21.33	3.00	15.33	24.33
72195	2.00	9.40	0.00	0.00	0.60
72196	8.00	11.67	0.00	0.00	0.33
72198	6.60	8.00	0.00	0.20	0.20
21001	3.00	8.60	0.00	0.00	0.80
21002	4.60	5.00	0.00	0.00	1.00
21004	8.60	8.00	0.00	0.00	2.80
21010	4.25	9.25	0.00	0.00	3.75
21105	10.40	31.20	0.00	0.00	3.00
46004	11.33	21.67	0.33	2.00	11.67
46005	5.60	15.00	0.00	0.60	3.20
46006	6.60	20.00	0.00	0.20	0.20
46007	19.20	27.40	2.00	4.40	11.80
46008	15.60	37.20	0.40	7.20	6.60
46009	8.67	15.00	0.00	0.67	0.00
46011	11.60	26.80	0.00	0.80	2.60
46012	2.20	4.20	0.00	0.00	0.00
46013	4.80	18.20	0.00	0.00	1.40

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Table B, continued

46014	10.00	18.60	0.00	0.00	3.20
46015	9.80	18.60	0.60	0.00	7.00
46016	3.20	9.20	0.00	0.00	1.20
46017	7.80	11.80	0.20	0.20	3.20
46018	4.50	6.50	0.00	0.00	2.00
46019	2.40	6.60	0.00	0.00	2.40
46020	12.60	28.40	0.20	3.00	7.00
46022	2.80	13.00	0.00	0.00	5.40
46024	3.00	17.67	0.00	0.00	9.00
46028	3.20	15.00	0.00	0.00	7.40
46029	11.40	27.20	1.60	0.00	15.80
46030	5.80	15.60	1.80	2.40	16.60
46031	22.60	27.00	5.60	1.40	28.40
46032	7.00	23.80	0.00	0.00	9.20
46033	4.60	15.40	1.20	0.00	13.60
46034	5.50	12.00	0.00	0.00	2.25
46046	2.20	3.80	0.00	0.00	5.00
46047	1.80	4.20	0.00	0.00	9.80
46054	10.67	31.67	0.00	2.00	22.67
46055	19.60	31.60	0.40	4.20	7.40
46110	5.20	7.40	0.00	0.00	0.80
46112	13.50	30.50	0.00	0.25	3.75
46121	1.80	11.60	0.00	0.00	2.20
46123	12.20	27.60	0.60	0.20	15.80
46125	5.67	16.67	1.00	0.33	4.67
46126	5.60	14.40	0.40	0.40	5.00
88002	7.60	8.00	0.00	0.00	2.00
88011	15.00	14.00	2.00	0.00	12.50
88017	0.50	4.75	0.00	0.25	0.00
88019	10.00	13.00	0.00	0.50	1.00
88020	10.67	9.67	0.00	1.00	3.67
88021	10.00	10.50	0.00	0.00	1.50
88022	8.25	4.75	0.25	0.00	2.75
88023	7.20	7.80	0.20	1.60	3.80
88024	5.80	10.40	0.00	0.00	4.80
88025	10.50	7.00	2.00	0.00	14.00
88026	10.75	12.50	0.50	0.00	7.25
88028	12.00	9.80	1.40	0.00	12.40
88031	7.50	11.00	0.00	0.00	20.00
88040	9.25	5.50	0.00	0.00	1.25
88046	11.75	18.50	0.25	0.00	12.25
88048	6.00	3.20	0.00	0.00	3.60
88051	4.20	7.20	0.00	0.00	1.80
88116	2.20	0.80	0.00	0.00	0.00
88127	11.50	11.50	4.25	0.00	19.50
88134	4.00	11.00	2.00	3.50	8.50

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Table B, continued

88900	8.75	23.75	2.75	13.75	36.00
88905	9.67	14.67	1.33	1.33	16.67
88910	2.00	3.33	2.33	0.00	7.67
88912	22.75	38.25	1.00	0.00	68.00
88914	11.50	12.75	4.25	0.25	45.25
88915	10.00	3.50	3.00	0.00	11.50
88918	21.50	12.50	5.50	1.50	26.50
90013	3.50	19.00	2.25	1.75	5.25
90015	4.00	28.40	1.80	6.40	7.00
90020	1.67	31.00	0.00	1.67	7.67
90021	2.00	30.50	0.00	2.00	5.00
90034	3.40	22.60	0.00	3.20	4.80
90035	4.50	25.00	1.00	3.75	4.00
90048	10.60	13.60	0.00	0.00	1.40
90051	1.50	3.00	0.00	0.00	0.00
90052	2.75	13.25	0.00	0.25	0.25
90053	6.75	9.25	3.00	4.00	9.50
90150	10.00	19.00	0.00	0.33	5.33
39004	19.80	4.60	0.00	0.00	0.20
39023	4.20	4.00	0.00	0.00	0.00
39028	17.00	15.80	0.00	0.00	16.20
39035	4.00	0.50	0.00	0.00	0.00
39137	4.67	0.00	0.00	0.00	0.00
63003	0.00	0.00	0.00	0.00	0.00
63012	15.20	15.60	2.00	2.20	9.80
63014	7.40	8.20	0.00	0.00	3.80
63015	13.00	11.60	0.00	2.00	7.80
63101	2.40	6.00	0.60	0.00	7.40
63119	2.00	5.00	0.00	0.00	3.20
63121	5.80	6.40	4.80	1.00	14.80
63126	2.00	10.60	0.00	0.00	0.40
63204	11.20	4.80	2.60	0.00	2.60
63205	2.60	1.60	0.00	0.00	1.80
63210	5.60	1.80	0.00	0.00	0.00
63220	1.00	16.67	0.00	1.67	5.33
63221	2.80	8.80	1.00	0.20	1.60
63225	7.40	4.20	0.00	0.00	2.00
63226	13.80	13.20	2.80	3.00	11.40
63906	1.50	4.50	15.00	1.00	27.75
63909	2.25	10.75	4.25	0.00	20.25
82003	13.00	6.00	0.00	0.00	0.00
82013	12.00	6.00	0.00	0.00	0.00
82019	7.33	1.33	0.00	0.00	0.00
82020	9.00	1.60	0.00	0.00	0.00
82022	8.00	2.00	0.00	0.00	0.00

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Table B, continued

88023	7.20	7.80	0.20	1.60	3.80
88024	5.80	10.40	0.00	0.00	4.80
88025	10.50	7.00	2.00	0.00	14.00
88026	10.75	12.50	0.50	0.00	7.25
88028	12.00	9.80	1.40	0.00	12.40
88031	7.50	11.00	0.00	0.00	20.00
88040	9.25	5.50	0.00	0.00	1.25
88046	11.75	18.50	0.25	0.00	12.25
88048	6.00	3.20	0.00	0.00	3.60
88051	4.20	7.20	0.00	0.00	1.80
88116	2.20	0.80	0.00	0.00	0.00
88127	11.50	11.50	4.25	0.00	19.50
88134	4.00	11.00	2.00	3.50	8.50
88900	8.75	23.75	2.75	13.75	36.00
88905	9.67	14.67	1.33	1.33	16.67
88910	2.00	3.33	2.33	0.00	7.67
88912	22.75	38.25	1.00	0.00	68.00
88914	11.50	12.75	4.25	0.25	45.25
88915	10.00	3.50	3.00	0.00	11.50
88918	21.50	12.50	5.50	1.50	26.50
90013	3.50	19.00	2.25	1.75	5.25
90015	4.00	28.40	1.80	6.40	7.00
90020	1.67	31.00	0.00	1.67	7.67
90021	2.00	30.50	0.00	2.00	5.00
90034	3.40	22.60	0.00	3.20	4.80
90035	4.50	25.00	1.00	3.75	4.00
90048	10.60	13.60	0.00	0.00	1.40
90051	1.50	3.00	0.00	0.00	0.00
90052	2.75	13.25	0.00	0.25	0.25
90053	6.75	9.25	3.00	4.00	9.50
90150	10.00	19.00	0.00	0.33	5.33
39004	19.80	4.60	0.00	0.00	0.20
39023	4.20	4.00	0.00	0.00	0.00
39028	17.00	15.80	0.00	0.00	16.20
39035	4.00	0.50	0.00	0.00	0.00
39137	4.67	0.00	0.00	0.00	0.00
63003	0.00	0.00	0.00	0.00	0.00
63012	15.20	15.60	2.00	2.20	9.80
63014	7.40	8.20	0.00	0.00	3.80
63015	13.00	11.60	0.00	2.00	7.80
63101	2.40	6.00	0.60	0.00	7.40
63119	2.00	5.00	0.00	0.00	3.20
63121	5.80	6.40	4.80	1.00	14.80
63126	2.00	10.60	0.00	0.00	0.40
63204	11.20	4.80	2.60	0.00	2.60

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Table B, continued

63205	2.60	1.60	0.00	0.00	1.80
63210	5.60	1.80	0.00	0.00	0.00
63220	1.00	16.67	0.00	1.67	5.33
63221	2.80	8.80	1.00	0.20	1.60
63225	7.40	4.20	0.00	0.00	2.00
63226	13.80	13.20	2.80	3.00	11.40
63906	1.50	4.50	15.00	1.00	27.75
63909	2.25	10.75	4.25	0.00	20.25
82003	13.00	6.00	0.00	0.00	0.00
82013	12.00	6.00	0.00	0.00	0.00
82019	7.33	1.33	0.00	0.00	0.00
82020	9.00	1.60	0.00	0.00	0.00
82022	8.00	2.00	0.00	0.00	0.00
82033	1.00	2.20	0.00	0.00	0.00
82038	1.25	0.25	0.25	0.00	0.00
82040	3.00	11.25	0.00	0.00	0.00
82042	0.60	18.20	2.80	0.00	15.80
82128	5.40	6.40	1.40	1.40	0.20
82135	0.00	3.60	1.00	0.00	10.80
82900	0.25	2.75	1.50	0.00	15.50
80002	5.75	2.25	0.00	0.00	0.00
80008	3.75	1.75	0.00	0.00	0.75
80101	8.40	10.40	0.00	0.00	0.00
80116	2.60	1.60	0.00	0.00	0.00
27004	0.40	0.20	0.00	0.00	0.00
27015	2.40	2.00	0.00	0.00	0.00
27020	3.20	11.00	0.20	0.00	0.00
27021	2.80	0.80	0.00	0.00	0.00
27022	2.50	6.00	0.00	0.00	0.00
27027	0.80	7.40	0.60	0.00	1.60
27032	13.00	9.20	1.20	0.00	0.40
27035	3.67	6.00	0.00	0.00	0.00
27036	8.50	4.00	0.50	0.00	0.00
27039	2.00	3.00	2.00	0.00	0.00
27044	0.40	4.60	0.00	0.00	0.00
27046	6.80	5.40	0.00	0.00	0.00
27111	0.33	2.00	0.00	0.00	0.00
27123	3.50	9.25	0.50	0.25	1.75
27156	1.50	4.50	0.00	0.00	0.00
27900	14.00	20.50	0.50	0.00	0.00
2001	7.00	6.40	0.00	0.00	0.00
2006	0.00	0.80	0.00	0.00	0.00
2013	8.33	8.00	1.00	0.00	0.00
2014	3.00	10.50	0.75	0.00	0.25
2017	1.00	4.60	0.00	0.00	0.00

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Table B, continued

2025	0.60	11.60	0.00	0.20	0.00
2026	2.25	9.50	0.00	0.00	0.00
2029	9.40	8.00	0.20	0.00	0.00
2031	0.40	2.20	0.00	0.00	0.00
2043	1.00	6.60	0.00	0.00	0.00
2047	3.00	7.20	0.00	2.20	0.00
2049	0.50	6.50	0.50	0.25	0.00
2050	3.40	7.00	0.00	0.00	0.00
2060	1.20	9.80	1.20	0.00	0.40
2061	6.33	14.33	0.00	0.00	0.00
2068	0.00	0.60	0.00	0.00	0.00
2071	3.50	0.00	0.00	0.00	0.00
2072	8.20	8.20	0.00	0.00	0.00
2102	16.40	3.00	0.00	0.20	0.20
2105	1.40	8.20	0.00	0.00	0.20
2111	3.00	6.40	2.40	0.00	0.20
2140	0.00	0.00	0.00	0.00	0.00
2203	3.00	2.50	0.00	0.00	0.00
2207	2.00	3.00	0.20	0.00	0.00
2211	8.60	10.20	0.00	0.20	0.00
2212	3.40	8.20	0.00	0.00	0.00
2217	0.00	0.00	0.00	0.00	0.00
51003	12.50	7.75	0.25	0.00	0.00
51030	0.00	3.20	0.00	0.00	0.00
51119	2.50	5.25	0.00	0.00	0.00
51229	0.20	0.60	0.00	0.00	0.00
25001	0.60	2.60	0.00	0.00	0.00
25006	5.25	0.25	0.00	0.00	0.00
25010	1.25	0.00	0.00	0.00	0.00
25017	0.00	0.00	0.00	0.00	0.00
25018	0.00	0.00	0.00	0.00	0.00
25019	0.00	1.33	0.00	0.00	0.00
25020	0.00	0.00	0.00	0.00	0.00
25022	0.00	0.00	0.00	0.00	0.00
25025	0.00	0.00	0.00	0.00	0.00
25026	0.00	0.00	0.00	0.00	0.00
25030	0.00	0.00	0.00	0.00	0.00
25031	0.00	0.00	0.00	0.00	0.00
25051	0.00	0.00	0.00	0.00	0.00
25052	0.00	1.60	0.00	0.00	0.00
25057	1.00	0.67	0.00	0.00	0.00
25060	0.00	0.00	0.00	0.00	0.00
25067	0.00	0.00	0.00	0.00	0.00
25068	0.00	0.00	0.00	0.00	0.00
25069	0.00	0.00	0.00	0.00	0.00

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Table B, continued

25070	0.00	0.00	0.00	0.00	0.00
25071	0.00	0.00	0.00	0.00	0.00
25074	0.00	0.00	0.00	0.00	0.00
25079	0.00	0.00	0.00	0.00	0.00
25085	0.00	0.00	0.00	0.00	0.00
25086	0.00	0.00	0.00	0.00	0.00
25087	0.00	0.00	0.00	0.00	0.00
25113	0.00	0.00	0.00	0.00	0.00
25116	0.00	0.00	0.00	0.00	0.00
25132	0.00	0.00	0.00	0.00	0.00
25175	0.00	0.00	0.00	0.00	0.00
25178	0.00	0.00	0.00	0.00	0.00
25903	0.00	0.00	0.00	0.00	0.00
25907	0.00	0.00	0.00	0.00	0.00
25910	0.00	0.00	0.00	0.00	0.00
25911	0.00	0.00	0.00	0.00	0.00
25912	0.00	0.00	0.00	0.00	0.00
25915	0.00	0.00	0.00	0.00	0.00
25916	0.00	0.00	0.00	0.00	0.00
25917	4.50	0.00	0.00	0.00	0.00
25918	0.00	0.00	0.00	0.00	0.00
25919	0.00	0.00	0.00	0.00	0.00
25920	0.00	0.00	0.00	0.00	0.00
25922	0.00	0.00	0.00	0.00	0.00

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