PROTECTED AREA NETWORKS IN AN URBANIZING LANDSCAPE: SPATIAL CHARACTERISTICS AND LAND ACQUISITION STRATEGIES

by

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ABSTRACT OF THE DISSERTATION

Protected Area Networks in an Urbanizing Landscape:
Spatial Characteristics and Land Acquisition Strategies

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The selection of land parcels for preservation and protection (i.e. the designation of a "Protected Area") is inherently a human social process engaging a complex suite of economic, political, and environmental factors. Since the 1960s, the U.S. federal government has encouraged local engagement in land conservation through new funding opportunities. Consequently, in some states a diverse collection of agencies, both public and private, have participated in the selection process. The following research examines whether diverse conservation organizations, sometimes acting with coordinated goals and sometimes acting independently, can collectively assemble a Protected Area network which aligns with some basic principles of biological conservation network design. The spatial patterns of one emerging Protected Forest (PF) network in the New Jersey Highlands are used a case study. This PF network consists of all forested habitat within the New Jersey Highlands Protected Areas. The primary finding are 1) although most large forest fragments have more than 80% of their land protected, medium and smaller-size fragments have less protection, 2) land cover change within 250 meters of PF boundaries

is highly variable and has both increased and decreased aspects of landscape permeability along those boundaries for forest species, and 3) land acquisition since 2000 has been proactive, relative to the threat of urban development. Because PA networks should represent and sustain regional biodiversity and ecosystem function, these findings have implications for future PA management. The pattern of protection of large habitat remnants in this region is favorable for sustaining existing ecological communities and processes. The increase in landscape permeability along the boundaries of some Protected Forests is also favorable because this facilitates species movements among protected habitat patches. However, because land acquisition has been highly proactive, the greatest amount of protection has occurred in the northern part of the region where urban development pressure is lower. The resulting uneven geographic distribution in this regional conservation network indicates that sustaining ecological forest communities and processes across the southern portion of the New Jersey Highlands may pose a significant future challenge.

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Dedication

To my boys, Henry and Tanner, you keep everything in perspective. You were toddlers when I started and are now both energetic young boys. I certainly admire those who finished their degrees in less time, but that could never compensate for all the Pinewood Derbies, kitchen dance parties, science projects, and family dinners I would have missed. Finishing this degree without missing your childhood is the most important milestone I could mark.

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Chapter 1: Introduction

Overview

The selection of land parcels for preservation and protection (i.e. the establishment of a "Protected Area") is inherently a human social process. It engages a complex suite of factors including economics, organizational and institutional capacity, political defensibility, land tenure, corruption, and donor regulation (Knight & Cowling 2007). Not all Protected Areas (PAs) are selected specifically for biological conservation; they exist to address a variety of resource conservation goals. However, this socially-driven selection process results in a spatial arrangement of Protected Area which inevitably has implications for sustaining biodiversity because protected status mitigates many types of anthropogenic habitat alteration. The following research examines how individual protected parcels selected by a variety of conservation organizations, sometimes acting with coordinated goals and sometimes acting independently, spatially aggregate across a regional landscape. It engages a question posed by Meyer et al. (2015) about whether such a decentralized and somewhat 'ad-hoc' social process driving land conservation can yield spatial patterns which align with some basic principles of biodiversity conservation network design. Unlike Meyer, I focus this question at a smaller spatial and temporal scale, examining the land conservation patterns emerging within a densely populated urbanizing region of New Jersey over the last 26 years.

Problem Statement: Land protection strategies and their pitfalls

There are known disparities in conservation land acquisition practices in the global north and global south. In the global south, multinational Non-Governmental Organizations (NGOs) often work with regional governments to establish conservation lands and parks through a centralized 'top down' selection process (West & Brockington 2006). One of the pitfalls in this practice is that areas with high biodiversity or endangered species are targeted for protection in a process which often overlooks, or directly conflicts with, existing user needs (Wells & McShane 2004; Naughton-Treves et al. 2005). In contrast, conservation acquisition in the global north is often highly deferential to the rights of property owners and local governments. The legal pathways for conservation land acquisition favor property owners and separate the development rights of land out from a multitude of other property use rights (e.g. mineral and water rights). As a consequence, the transfer of land into protected status often depends upon a 'willing-seller, willing-buyers' model (Fairfax et al. 2005). The reliance upon willing sellers shifts the power for selecting Protected Areas into a more decentralized and 'bottom-up' structure.

There are several reasons such a decentralized land protection process could, in aggregate, yield PA networks which align with general biological conservation network design principles. 1) Certain basic conservation network design principles, such as maximizing size and connectivity among protected lands, are simple to grasp and have been well articulated since the 1970s. In the U.S. there is evidence that municipal planners and even developers (Beuschel & Rudel 2009) have integrated such guidelines into development practice. 2) Local governments, citizens, and non-profits often have extensive formal and informal social networks which enable them to identify and act quickly upon land conservation opportunities. This is an extremely important factor both in areas where desirable conservation lands are held privately, and in areas where less formal land claims hamper federal or state-level acquisition action (e.g. Harrison, 2006)

Many conservation opportunities would undoubtedly be missed if local groups had to solicit the involvement and approval of state, provincial, or federal agencies.

However, a decentralized land selection system reliant on willing sellers also has the potential to produce a Protected Area network that would not align well with conservation network design principles for several reasons. First, land is acquired along property boundaries rather than ecological habitat boundaries. This increases the likelihood that contiguous habitat may not have contiguous protection. Second, land conservation done with a local rather than a regional focus could produce a network of many small PFs, isolated from one another by an urban or agricultural matrix. Third, there is potential for significant spatial mismatch between the location of groups willing and able to fund local land protection, and the highest priority lands for resource conservation.

The purpose of this research is to assess one emerging Protected Area network by contextualizing spatial patterns of PAs habitats within the landscapes around them. Protected Area (PA) acquisition in New Jersey was devolved from the federal to the local level beginning in the 1960s (Foresta 1981). This trend toward local actors selecting Protected Areas has been reinforced through a proliferation of ballot measures in more recent decades (Heintzelman et al. 2013). As a result, New Jersey offers a unique location to study the spatial patterns of a PA network in which a significant amount of land is selected through a decentralized "bottom-up" process. These spatial patterns have biological implications. If small, isolated patches of protected habitat proliferate across the region because actors focus on protecting their proverbial 'backyards', this can impact the numbers and types of species which persist locally and regionally (Pickett & Thompson 1978; Newmark 1995). If such patterns are emerging, policies which encourage a decentralized land selection/protection process may not be desirable means to an end for supporting biodiversity protection. Making links between social and biological process across landscapes that host Protected Area networks is therefore important for guiding future policy on land protection.

Structure of this Dissertation

At the most basic level this work asks the question, "Which land parcels get protection and how are they spatially arranged in the greater landscape?" Chapter 1 describes the federal, state, and local legislative structures which make land conservation possible, along with some general principles guiding the design of land networks for biological conservation. I use NJ as a case study for the spatial arrangement of land conservation networks where the acquisition process is largely decentralized and the land use has been fragmented by urbanization (Foresta 1981). The output of this socially-based selection process is fundamentally just a map, a spatial portfolio of all Protected Areas compiled from several sources, the development of which I describe in Chapter 2. Chapters 3, 4, and 5 analyze this map in the context of local and regional land use. Chapter 3 analyzes the arrangement of protected forests within larger contiguous forest fragments. Chapter 4 analyzes the land use and land change on protected forest borders of varying sizes. Chapter 5 analyzes strategies of protection and probabilities of protection for land of different resource value. Taken together, these analyses offer insight into the potential benefits and obstacles 'bottom-up' land conservation offers for supporting biological conservation.

Background: Multi-level Government Legislation for Land Protection

Federal-Level Land Protection Legislation: The Weeks Act of 1911, The Land and Water Conservation Fund of 1964

Land protection for resource conservation and recreation within the eastern U.S. has been a challenge since the late 1800s. As the National Parks and National Forests in the West grew in numbers and acreage, few comparable public lands were designated in the East (Clawson 1983). This disparity occurred because even though federally-owned lands in the West could be retained for public use, most land in the East had already passed into private ownership decades before. For land conservation to proceed in the East, most would have to be purchased. Throughout the 1800s, the federal government had no legal authority to purchase land for recreation or resource protection (Fairfax et al. 2005).

However, the extent of damage done to eastern forests through extractive industries like timber and mining eventually opened a pathway for acquisition. Since the federal government could regulate interstate commerce, it could purchase and protect forests in the headwaters of a stream to prevent siltation that might compromise the navigability of the waterways. The Weeks Act of 1911 codified this federal government purchasing power and extended the power to provide subsidies to industry and local governments to promote protectionist practices. The Weeks Act is the legislative cornerstone of all federal acquisition programs funded today, including the Land and Water Conservation Fund (Fairfax et al. 2005).

Roughly 50 years later (1964), Congress passed the Land and Water Conservation Fund Act (LWCF) (National Park Service 2016a). This legislation was the federal response to the residential suburban buildup of the 1950s, particularly apparent in the growing 'megalopolis' along the Atlantic Coast from Boston to Washington D.C. (Foresta 1981). In the post WWII era, sportsmen, naturalists, and urban residents raised protests over land development. This prompted Congress to form the Outdoor Recreation Resources Review Commission (ORRRC) to address population growth and recreational needs into the next century. The commission's report identified a national gap between projected need and available "open space" (i.e. public land for recreation and natural resource protection), and raised particular concerns about the lack of public space along the country's northeastern coast.

Since its inception, the LWCF has been one of the most important and consistent sources of funding for land acquisition throughout the U.S. Originally set as a 50 year funding provision, the popular LWCF was recently extended until 2018 (National Park Service 2016a). Funding for the entire program began at $100 million per year in 1965, but was raised to $300 million through the 1970s and 1980s. Since 1999, the actual appropriations have ranged from $149 to $573 million. Among all permanent land protection programs funded by the federal government, the LWCF was the top spender from 1992-2001 (Lerner et al. 2007). Under the umbrella of providing and protecting open space, throughout its lifetime, the LWCF focus has expanded beyond outdoor recreation and to include a variety of ecological land conservation goals (e.g. provisioning of ecosystem services like clean water, and habitat for biodiversity).

There are two funding arms of the LWCF: the Federal Land Protection Program and the State Assistance Program. The first arm funds direct federal acquisition of land. The second provides funds to states and local communities though 50:50 matching grants. Specific parcel acquisitions can be proposed by any state or local agencies willing to provide the matching funds, provided that their state has prepared and revised a Statewide Comprehensive Outdoor Recreation Plan (SCORP) every five years.

The LWCF matching funds were deliberately structured to devolve responsibility of open space acquisition to more local levels for several reasons. First, with skyrocketing land prices, there was a sense of urgency about acquiring private lands as early as the 1960s (Foresta 1981). By encouraging local participation in the selection and ownership of protected lands, the federal government could reduce local resistance to acquisitions by deferring to traditions of home rule and state's rights. Second, the matching grants stretched federal dollars and provided a means to fund acquisitions in high priority, densely population metropolitan regions without forcing rural taxpayers to foot the entire bill (National Park Service 2016a). These high density regions included the 'megalopolis' along the northeast coast, a site where the largest gap between available and projected needs for public recreation lands existed in the 1960s. This was also the region where most of the undeveloped land was held by private, municipal, or state interests (Fairfax et al. 2005).

Through LWCF monies, since 1965 approximately 2.2 million acres have been directly purchased by the federal government and added to the National Parks System. An additional 2.6 million acres have been acquired through the matching LWCF funds distributed to state, municipal, and local organizations (The Land and Water Conservation Fund Coalition 2015). In New Jersey, $207 million LWCF dollars have been spent since 1965 through the Federal Land Protection Program, and $119 million have been spent through the State Matching Grant Program. Forest Legacy and Habitat Conservation Grants together total an additional $20 million (The Land and Water Conservation Fund Coalition 2018). These figures on the matching grants are an indication of the significant contributions local organizations have been making to the open space selection process both across the country and specifically within New Jersey.

State-Level Land Protection Legislation: the NJ Green Acres Program

The aforementioned ORRRC report produced in 1961 put considerable pressure on each state to "develop a long-range plan for outdoor recreation, to provide adequate opportunities to the public, [and] to acquire additional areas where necessary" (National Park Service 2016b). A subsequent inventory of open space within the New York metropolitan area revealed that New Jersey was behind all other states in providing open space for residents. Whereas the National Parks and Recreation Association recommended a minimum 1 acre of open space per 1000 residents for smaller municipalities (i.e. boros), and 5 acres per 1000 residents for larger municipalities, New Jersey only averaged 0.3 acres and 0.4 acres, respectively (Foresta 1981). New Jersey citizens responded to this report by passing a 1961 ballot measure dedicating $60 million for land acquisition. This was the beginning of the New Jersey "Green Acres Program". It connected state projects to the newly established federal LWCF dollars through the program's Bureau of Planning and Information Management. Because it was created in response to and in conjunction with the LWCF program, Green Acres mirrors the LWCF structure. It has two funding arms: one for acquisitions by the state and one for matching grants and low interest loans provided to municipalities and non-profits to support land acquisitions they propose. Like the LWCF, it has evolved through time to support open space acquisition for a variety of resource protection goals.

By design, the Green Acres funding was intended to support a more locally-based selection process for open space. However, social response to the program also enhanced the role of local decision making beyond what might have been intended. Foresta's (1981) research into the first decades of the Green Acres Program offers detailed analysis of the social and administrative conditions which pushed NJ toward a more decentralized and user-controlled land selection process. The acquisition of state-owned lands was effectively decided by two state offices: the Office of Fish & Game, and the Office of Parks and Forests. Acquisition of municipal and county-owned lands through the matching grant program was originally also to go through these the two state offices before gaining approval. In this way, decision making would be centralized and the state could exercise considerable control over the selection of open space.

In practice, this vetting process at the state level never occurred. State and federal administrators badly misread local interest in open space protection and local government ability to provide matching funds. Despite the urgency of the ORRRC report and the willingness of NJ citizens to fund open space, the anticipated 'latent demand' for funding was weak or non-existent at the local level. In the first 12 years, this "sluggish demand brought pressure on local administrators to spend money quickly, first to show that there really was an open space crisis, then to show that they were doing something about it." Consequently, the state was anxious to approve any municipal application for matching funds, a condition which put the applicant in greater control of open space selection. Foresta concludes that

> *… the configuration of the program's results is not due in any measure to a specific state distribution policy; rather it is the result of hundreds of local decisions that, having been made, trigger state spending and thus shape the program. The legacy of this has been the uneven distribution of local open space acquisitions, acquisitions largely determined by the administrative capacity of elected officials to take advantage of the Green Acres program.*

Despite this early unintended reorganization of acquisition responsibility (or perhaps because of it) the Green Acres Program gained popularity during its first funding period. Twelve subsequent NJ state ballot measures to renew the program were passed from 1961 until 2009, totaling 3.2 billion dollars (Heintzelman et al. 2013). The largest amount ($250 million) was appropriated in 1995, after which there was a decline in funding until 2009 when $240 million were appropriated for both Green Acres and the Blue Acres flood prevention program (NJDEP, 2014). It is now one of the nation's oldest and most consistently funded state open space acquisition programs. The NJ Green Acres Program has outpaced the LWCF spending in New Jersey, expending $383 million directly on acquisitions in New Jersey since its inception in 1961 (figures calculated from data provided by Eric Knudson of the Green Acres Program). At the local level, funding for open space has been further magnified (and control of open space selection has been further consolidated) because many New Jersey municipalities have passed ballot measures similar to those which fund Green Acres.

Local-Level Land Protection Legislation: County, Municipal, and Special District Ballot Measures

Across the U.S. in recent decades, ballot measures like the one which initiated the NJ Green Acres Program have become increasingly popular tools for funding open space protection at the municipal, county, and state level. Also called 'voter referenda', these ballot measures appropriate funds through voter approval of a ballot question which proposes the amount and source of the funds (e.g. a tax or bond) and the use of those monies (Myers 1999; Kotchen & Powers 2006). Use of monies typically include land acquisition, program administration, and/or land management.

Since 1988, 1740 of 2299 (~76%) of U.S. voter referenda have passed, appropriating over $56 billion for land preservation. These funds are often central to leveraging matching funds from state, federal, and NGOs. From 2000-2004 alone, 711 of 938 ballot referenda conducted at state, county, and municipal levels passed, authorizing 15.6 billion dollars for the acquisition of open space and development rights (Heintzelman et al. 2013). This is more than the monies allocated to the largest federal land protection program in the U.S. (the Conservation Reserve Program or CRP) during the same period (Nelson et al. 2007). Whereas the CRP funds are spent on short-term land rentals which offer no long-term protection for land (Lerner et al. 2007) funds derived from ballot initiatives generally go directly into permanent protection through acquisitions and easements.

The proliferation of ballot measures across NJ mark the state as a leader for local-level engagement in land conservation (Myers 1999; Solecki et al. 2004; Lerner et al. 2007; Szabo 2007). From 1996-2004, New Jersey appropriations at the state and county level contributed disproportionately high amounts to overall U.S. tallies of voter-approved funding. New Jersey municipal ballots from 2000-2004 alone allocated $530 million for open space acquisition. This is roughly 14% of the $3829 million dollars appropriated across all U.S. municipalities during that same time period (Nelson et al. 2007). Although not all these funds are spend directly on acquisitions costs, the scale of appropriations makes it clear that local governments in New Jersey play a significant role in open space selection there. Local commitment to open space is also manifest in the preservation of township lands, a fact which is not reflected in monetary accounting because townships do not pay to acquire their own land. Like the federal government in the 1800s (Fairfax et al. 2005), they simply legislate the retention of lands for open space rather than selling them to private entities. Local leadership in protected area selection is further evident via the number of non-profit organizations which facilitate open space protection in specific regions (New Jersey Conservation Foundation 2015).

Szabo (2007) states that the data on open-space referenda tell one of the "great conservation stories of the last decade". Part of that story is the engagement of municipalities. Although the largest amount of money has been generated by states and counties, the greatest number of measures have passed at the municipal level. He suggests citizens have been motived to increase their engagement with local protection efforts because “unfavorable political and fiscal dynamics at the federal level, which are likely to persist for some time, suggest that he heavy lifting will not be done in Washington”. However, the demographics of the communities most engaged in this endeavor have long raised suspicion that open space protection is wielded as a tool to promote ‘exclusionary zoning’ more than ecological protection (Foresta 1981; Duncan & Duncan 2001). A quick glance at any map of ballot measures reveals the uneven geographic distribution of these pockets of open space advocacy (The Trust for Public Land 2016).

Published analysis provides evidence to substantiate certain links between the socioeconomic characteristics of a community and the level of support for open space funding measures. In addition to NJ, ballot measures have been particularly successful in the states of Florida, California, and Massachusetts, proliferating at multiple levels of government (Lerner et al. 2007). Analysis of California communities in the 1970s showed that voting for conservation increased with level of education and decreased with the share of construction-related unemployment within a county (Deacon and Shapiro, 1975 cited in Heintzelman et al., 2013). A study by Kotchen and Powers (2006) which analyzed data from both New Jersey and Massachusetts (1998-2003) found that voter referenda are held more often in wealthier communities with greater levels of population growth. These results are largely consistent with a study in NJ focused specifically on the 1998 state referendum, which found that higher average incomes predicted voter approval by municipality (Solecki et al. 2004). Such studies lend support to suggestions that open space protection, particularly at the local level, is sometimes invoked to exclude certain types of development and 'defend' local landscapes (Rudel 2013).

However, socioeconomic composition of communities rarely stands alone as the most important predictor of referenda success. Virtually all authors cited above found significant effects of other social and environmental factors on the approval of ballot measures. Kotchen and Powers report mixed impacts of some factors. In their analysis, more open space loss in years prior to a ballot measure actually reduced voter support at certain scales. Solecki et al. found that although higher average income predicted voter approval, higher amounts of existing conserved land, and high rates of rural-to-urban land conversion were also correlated with lower voter approval at the municipal level. Nelson et al. (2007) found that open space loss and infrastructure growth, as well as higher levels of employment, increased the likelihood of success for open space referenda at the municipal level from 2000-2004. In that study, higher voter education level and the condition of no additional tax burden from the open space approval also increased the likelihood of success.

These studies on voter referenda offer insight into the complex human social patterns which directly and indirectly shape the landscape patterns of Protected Area networks. For example, wealthy communities may not always coincide with the land which has the highest resource value, yet they may be the communities most willing and able to fund local land protection. As a consequence, the size and spatial patterns of Protected Areas will be uneven, particularly in regions where income, the pace of urban development, and prior land protection are uneven.

Natural Reserve Network Design Theory

Links between open space and ecological conservation

Although a sensible approach to increasing access to open space, many of the aforementioned legislative mechanisms were not originally intended to address the 'global biodiversity crisis' which would be articulated in the ecological literature of the 1970's and 80s (Soule 1985; Meine et al. 2006). This call for biodiversity protection expanded the expectations of lands protected through open space programs from predominantly serving people to also representing and protecting biological diversity (Prendergast et al. 1999; Margules 2000). The introduction of the 1973 Endangered Species Act formally codified that expectation by providing a legal mechanism to protect species. By the time biodiversity conservation became a rallying cry for more open space protection, many funding structure, such as the LWCF and NJ Green Acres, were well established. Rather than legislate new funding, many existing programs were adapted and adopted to accommodate biodiversity protection.

This re-purposing of open space funds, and re-imagining of open space purposes created some obvious challenges. Open space protection for recreation or provisioning of navigable waterways need not always be large, well connected, or representative of species in order to serve human needs. In contrast, habitat size, habitat connectivity, and native species all figure prominently in debates about conserving ecological communities (Simberloff & Abele 1976; Pickett & Thompson 1978; Grumbine 1990; Parrish et al. 2003; Jenkins et al. 2015). Land acquisition focused specifically on representing and sustaining biodiversity is known as "systematic conservation planning" (Margules 2000). Systematic conservation planning yields "natural reserve networks", or "biodiversity networks". These reserve lands are an explicit sub-category of Protected Area recognized within IUCN categories I-IV (Chape et al. 2003; Dudley 2008).

There is a tension between the more general practice of establishing PAs and the practice of systematic conservation planning. Networks of Protected Areas are not necessarily biodiversity reserve networks. Therefore, portions of any PA networks may not have specific goals for sustaining biodiversity. Nonetheless, every Protected Area arguably contributes to biodiversity protection in a land conservation network by restricting anthropogenic habitat alteration. For this reason, the spatial characteristics of PA networks and their relationship to surrounding landscapes are highly relevant to biological conservation. Consequently, it is important to examine how PAs networks follow, or diverge from, spatial arrangement favored in conservation network design.

Reserve Network Design Theory and Systematic Conservation Planning

In its infancy, theory about the design of natural reserve networks drew upon MacArthur and Wilson's Island Biogeography Theory (IBT) (1967) to predict how spatial size (i.e. area) and isolation (i.e. distance to similar habitat) of a nature reserve might affect the species diversity and abundance it could support (Diamond 1975; Simberloff & Abele 1976; Prendergast et al. 1999). Anthropogenic land conversions appeared to produce patches or 'islands' of habitat in 'seas' of human land use (e.g. agriculture or residential development) (Haila 2002). Sustaining biodiversity through selective protection of these emerging habitat 'islands' seemed as though it might be a predictable exercise if IBT were applied (Diamond 1975; Diamond et al. 1976)

For a variety of reasons, Island Biogeographic theory falls short when predictions are tested against ecological patterns in fragmented terrestrial habitats (Laurance 2008). Terrestrial landscapes produce novel dynamics which are not accounted for in IBT because the habitat loss and habitat alteration defining the remnant habitat patches is not uniformly distributed across landscapes. Consequently the land use/land cover (a.k.a. the "matrix") surrounding each habitat remnant are not uniformly 'hostile' to all terrestrial organisms. Rather, they have varying levels of permeability and resources for individual species (Forman 1995; Prevedello & Vieira 2010). Such novel ecosystem dynamics make it difficult to apply IBT to selection of parcels for biodiversity protection.

Novel human social dynamics are also not accounted for within IBT. The selection of land parcels for protection is often constrained by the practical reality of real estate acquisition practices. Deference to property owner's rights means that land protection occurs along property boundaries rather than habitat boundaries. Consequently, it is logistically difficult to protect an entire habitat remnant along the land cover boundaries. Because land use is superimposed upon land cover, the two boundaries do not necessarily coincide. This also makes it difficult to translate IBT theory directly to natural reserve network design and management. All this is not to suggest that the effect of parcel size and isolation emphasized by IBT are useless metrics. Indeed, it would be difficult to think about any kind of open space or reserve network in the absence of such information. However, they are starting points in current practices of reserve network assembly, rather than endpoints.

Facilitated by increased computing power in the 1990s, complex algorithms now use data-driven approaches to target parcels for inclusion in biodiversity networks. These algorithms recognize that reserve network design depends upon the species in question *and* the dynamics of land use in a region, not merely protected habitat fragment size and isolation. Three key concepts guide the design of computer-based planning tools: complementarity, irreplaceability, and vulnerability (Sarkar et al. 2006).

Complementarity aims to select sites which maximize differences in biota. Irreplaceability quantifies the probability that a site must be included in a network design solution in order to achieve biodiversity targets. Vulnerability considers the probability of biota persisting at a given site (see also Prendergast et al., 1999 for a brief history of site selection algorithms). Vulnerability at a site depends not only on the quality of the site itself, but also the context of that site within the broader ecological and human landscape.

The benefits of such planning exercises are multiple (Sarkar et al. 2006). However, there are also drawbacks. They require not only an abundance of data, but also specialized skills to work with and interpret results from the many software packages available. These factors, among others, contribute to the gap between research and implementation of systematic planning in conservation land selection (Prendergast et al. 1999; Knight et al. 2008). In the absence of the full suite of data and skills needed for biodiversity network planning, practitioner involved in land acquisition may default to simple guiding principles of natural reserve network design. These are outlined in many conservation textbooks (e.g. Primack, 2010) and have percolated beyond the conservation community into decision making by municipal planners and developers (Beuschel & Rudel 2009). Among these principles are choosing (when possible) parcels for conservation which are larger, encompass entire habitats, and have some spatial connectivity to similar habitat patches across the landscape.

Objectives

I argued earlier in this chapter that NJ is a state with a decentralized system of land acquisition, and a history of land acquisition pre-dating many of the current goals and tools associated with biodiversity protection. The PA network there, as a whole, has not been systematically planned, although efforts at regional planning have been strong since the 1970s in the Pinelands (NJ State Pinelands Commission 2015), and the 1990s in the Highlands (Phelps & Hoppe 2002). This study will focus on the Highlands. Because of the piecemeal acquisition process resulting from limits on 'available' land for conservation, the current PA network there does not constitute a formal biodiversity reserve network. Although, subsets of the land system may serve that role for some species. The question I pose is whether this network of Protected Areas has, through time and through the decisions of multiple actors, spatially aggregated into patterns beneficial for biodiversity.

Expanding beyond the patch-based framework often applied to PAs (Fahrig, 2013), I approach my analysis from a landscape-perspective, linking aspects of PA vulnerability to their spatial context in a regional human-dominated landscapes (Wiens 2009). The importance of integrating landscape context into ecological conservation and habitat restoration is a common theme across many disciplines (Wells & McShane 2004; Naughton-Treves et al. 2005; Cummings 2015; Hauck et al. 2016) as well as a priority in restoration policy (Laestadius et al. 2015). In an era of earth's history where most biomes are dominated by human land use (Ellis & Ramankutty 2008), species populations and habitats of conservation interest necessarily occur within and are adjacent to these dynamic landscapes. Both social and ecological processes present within protected and unprotected habitats feed back on one another (Wiens 2009). Thus, PA function and PA community sustainability is inextricably tied to the broader landscape. Only through expanding to a landscape-level analysis can one begin to identify strengths and challenges at individual PAs and across the PA network.

I analyze three spatial patterns in the subsequent chapters of this dissertation. Two are spatial guidelines published by Shafer (1997) which expand upon ideas originally presented by Diamond (1975). They are reproduced in various forms in many basic conservation textbooks (Figure 1). As guidelines, they greatly over-simplify the problem of conservation, offering only comparative "better" or "worse" designs for biological conservation networks. However they are useful for indexing trends in emerging networks because they allow comparisons between patterns of protected and unprotected habitat. (For example, one can analyze whether the largest intact habitat fragments in an ecoregion are protected or unprotected.) I work under the assumption that most actors engaged in open space acquisition have working knowledge of these principles and try to choose the "better" option when possible. The third aggregate pattern I analyze attempts to characterize the regional strategy for land acquisition in my study site over the last decade. A brief overview of each analytic chapter follows.

First, it is preferable to protect a large parcel over a small one and preferable to protect an entire habitat rather than a portion of it (Principles A & B, Figure 1). In Chapter 3, I compare the patch size and percent protection of all remnant forest habitat within my study site. I discuss the results in the context of the broader landscape and the interaction between land use and land cover. Specifically I consider the geographic distribution of remnant forest fragments of differing sizes, and how large and small protected habitats are further distributed within those remnant forests.

Second, it is preferable to reduce habitat isolation around a PA (Principles E and F, Figure 1). In Chapter 4 I consider the question of isolation from the perspective of Shafer's suggested guideline that conservation parcels should have a 'permeable' edge rather than an edge with an abrupt land cover change (Shafer 1997). My definition of permeability follows is drawn from a review paper by Prevedello and Vieira (2010). I analyze the land cover at the borders of protected forests to determine if protected patches of different sizes have different border permeability. I consider the results in the context of landscape change through time around protected forest patches.

Third, drawing on objectives within systematic conservation planning, I explore evidence for two regional patterns in PA selection in Chapter 5. First, I analyze the probability of protection for sites with high resource value and high likelihood of habitat conversion, versus those with high resource value and low likelihood of land conversion. Second, I analyze how socio economic and development patterns in municipalities influence regional land acquisition. Independently and collectively, the patterns I analyze have different implications for sustaining wildlife populations within Protected Areas. Acquiring sites with low threat of conversion means that ecological communities may be lost in those areas which overlap with high human demand. However, favoring threatened sites for acquisition may result in PAs which are small and isolation (Norris & Harper 2004; Spring et al. 2007).

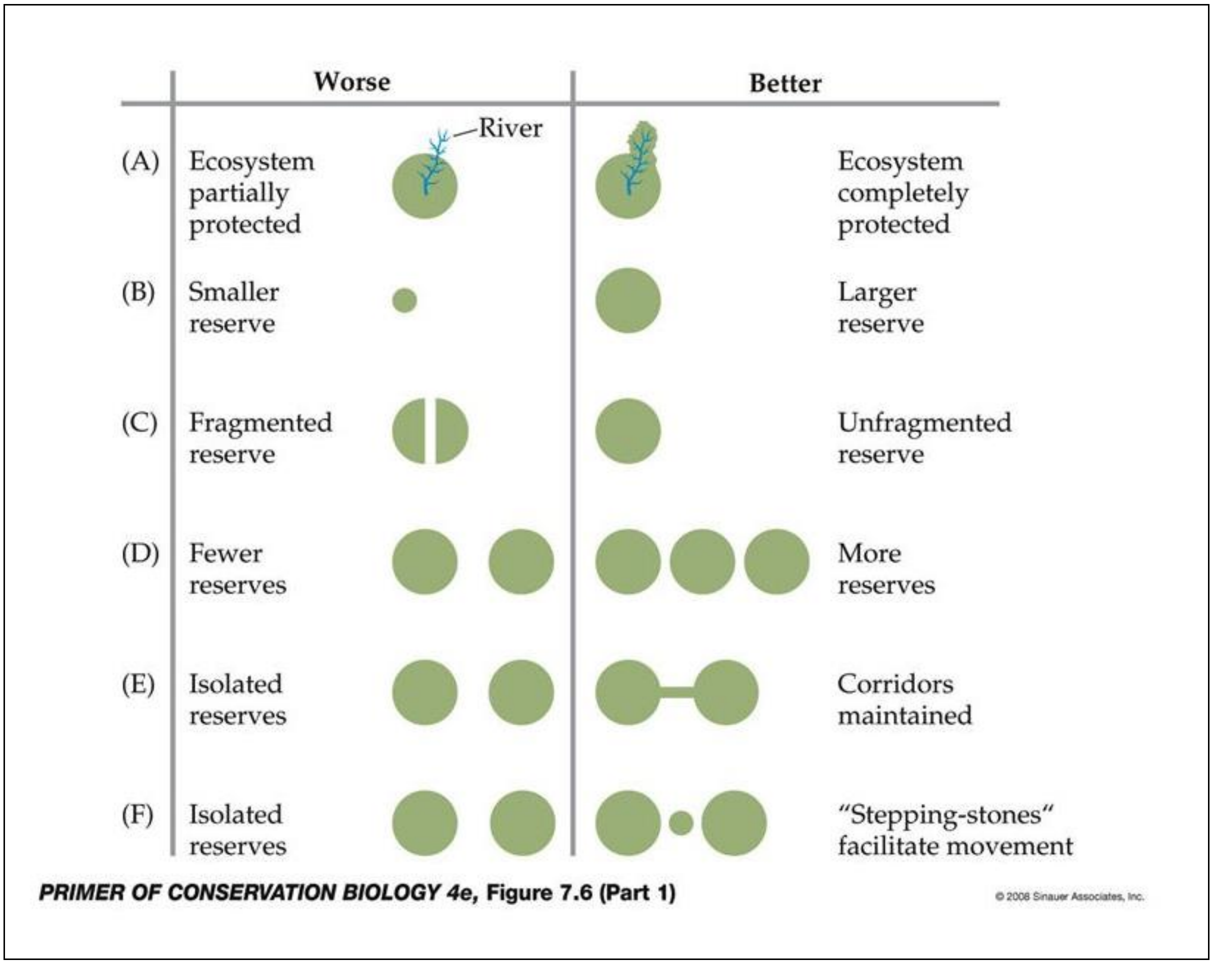


Figure 1 A subset of several Natural Reserve design guidelines for terrestrial ecosystems derived from island biogeography theory (Source: Primack, 2008)

Chapter 2: Methods - Study Site and Database Development

Study Area

My study area covers the 3477 km^2 (1,343 mi^2) physiographic region of New Jersey known as "the Highlands" (Phelps & Hoppe 2002). The NJ Highlands are part of the larger "Northeastern Highlands" ecoregion (USGS, 2016). The ecoregion extends continuously across portions of Pennsylvania (PA), New Jersey (NJ), New York (NY), Connecticut (CT), Massachusetts (MA), Vermont (VT), New Hampshire (NH), and Maine (ME), but also has noncontiguous patches in New York's Adirondack and Catskill mountain region (Figure 2.1). Ecoregions are defined by integrating data on climate, geology, physiography, soils, vegetation, hydrology, and human factors. Their classification "reflects patterns of land-cover and land-use potential that correlate with patterns visible in remotely sensed data" (USGS, 2016). The Northeastern Highlands ecoregion is characterized by relatively nutrient poor soil, a gradient of low to high mountains, sparse human population, and a history of glaciation manifest in its rocky soils, wetlands, and lakes (USGS, 2016b).

Land Use History

Temperate forests once dominated the entire Northeastern Highlands ecoregion. Although much of this original forest cover was lost to agriculture and extractive industries in the post-colonial period, a period of forest recovery ensued in the mid to late 1800s as landowners abandoned these land use practices (Ramankutty et al. 2006; Ellis et al. 2010). However, since the 1970s, forests in the southern end of the ecoregion have experienced a net loss largely to residential and commercial development (Drummond & Loveland 2010).

Like the greater Northeastern Highlands region in which it is embedded, the New Jersey Highlands experienced extensive deforestation from the mid-1700s until the late 1800s. Forests were cut to fuel the iron ore industry in the northern Highlands and to increase agricultural production in the southern Highlands (Kury & Wacker 2011; Lathrop 2011a). The opening of rich agricultural land in the mid-western U.S., coupled with regional shifts in industrial production, reduced deforestation throughout the early decades of the 1900s. Forests experienced regrowth until roughly the 1940s. At this point, the region's proximity to New York City and infrastructure investment (first into roads, and later into passenger rail networks) made the NJ Highlands desirable real estate for peri-urban and suburban residential and commercial development (Shutkin 2000). By 2000, 24% of the NJ Highlands was classified as 'developed land' (a classification which in the context of this study includes residential, commercial, and industrial land use at all densities). At the turn of this century, only ~51% of the land remained in forests and 7% in wetlands (Phelps & Hoppe 2002).

Land Conservation in the Highlands

The bedrock, topography, and rainfall of the Northeastern Highlands all contribute to it being a region with abundant amounts of potable, accessible freshwater. The region's provision of clean water for population centers along the eastern seaboard has been recognized since the 1890s when the NY Adirondack Park was first established for the protection of similar ecosystem services (Jacoby 2001; Fairfax et al. 2005). In the 1990s, New York City began investing in forest conservation throughout the Catskill mountain portion in order to secure the future of the city's present drinking water supply (Chichilnisky & Heal 1998). More recently this dependence on potable water from the Highlands has been recognized via several pieces of conservation legislation. In 2004, the federal government passed The Highlands Conservation Act (108 U.S.C. § 2375) aimed at protecting forests near the urban centers of the northeast. This targets the states of PA, NJ, NY, and CT (Figure 2.2).

In conjunction with the Federal Highlands Act, New Jersey passed its own legislation (The Highlands Water Protection and Planning Act of 2004, NJSA 13:20-1). Because ecological habitat crosses administrative boundaries, effective drinking water protection requires coordinated conservation at a watershed scale (Tarlock 1986; Lathrop et al. 2007; Qiu et al. 2014). NJ, with its strong tradition of land control at the municipal level, had no mechanism for coordination planning for regional resources protection. The NJ Highlands Act promotes regional planning across an area in which 88 municipalities and seven counties have jurisdiction. The objective is to mitigate urban development impacts and to hold all municipalities accountable for contributing to water quality protection in the region. The NJ Highlands Act assigned approximately 1600 km^2 of the Highlands to a "Preservation Area" and the remaining land to a "Planning Area" (Figure 2.3). Development in the Preservation Area is subject to a permitting process which determines the project's consistency with regional resource protection goals. It is important to note that this goal of coordinated planning is recent. Much of the urban development and conservation lands within the NJ Highlands pre-date this legislation.

Since the 1960s, New Jersey has legislated funding for significant amounts of open space acquisition through both state (Foresta 1981) and local ballot measures (Kotchen & Powers 2006; Lerner et al. 2007; Heintzelman et al. 2013). The state also supports a variety of federal and local conservation programs for private owners (NJAS, 2006). Consequently, a significant portion of the Highlands now has protected status. The combination of regional conservation legislation and locally-focused conservation spending has brought the Protected Area network there closer to a final configuration than most regions elsewhere in the U.S. There simply is less land available to either protect or develop. Thus, it is one of the most appropriate sites to study the spatial patterns of Protected Areas emerging from a decentralized process for selecting conservation lands. It can provide insight into how current NJ land acquisition policies 1) create patterns of partial protection within habitats, 2) juxtaposition PAs within other land use 3) encourage either proactive or reactive land acquisition (Spring et al. 2007).

Protected Area Data Sources

My study units are the Protected Areas (PAs) throughout the New Jersey Highlands. Within these, I focus on the forested portion, a unit of study which I will refer to as upland "Protected Forests" (PFs) throughout the body of this text to distinguish my analytic units from more general reference to conserved forest land. In the context of my study, my use of the term 'protected' includes lands acquired through purchase or donation for conservation purposes, as well as those enrolled in land conservation programs, or those with conservation easements. These lands may or may not overlap with the stricter IUCN designations (Chape et al. 2003), but all contribute to sustaining regional biodiversity by protecting habitat. I identified the spatial location and extent of these units within the NJ Highlands by integrating four GIS datasets: 1) the New Jersey Highlands Council dataset of Preserved Lands (HCPL), 2) the United States Geological Survey's Protected Areas Database (PADUS), 3) the New Jersey Farmland Preservation Program dataset (FPP), and 4) the New Jersey Department of Environmental Protection's dataset of State, County, and Federal lands (NJSCF).

The HCPL data were last compiled in 2011 under the guidance of the New Jersey Highlands Council and include federal, state, county, local, and non-profit lands with protected status. The location and status of all protected parcels in the database has been verified by participating municipalities through the Plan Conformance process (New Jersey Highlands Council 2011). This database provides only spatial data on protected parcels and a single attribute indicating ownership status at the time the data were compiled (e.g. state, federal, municipal, non-profit).

PADUS includes lands purchased outright for conservation and private lands protected through conservation easements. It was compiled from 2005-2011 through the joint efforts of federal agencies and some of the larger U.S. non-profit land trusts (USGS, 2012). It includes all Protected Area data available to participating agencies as of 2011 and up to 28 attributes on each spatial location.

The FPP data (NJDA-SADC, 2014) were updated in 2014 and include agricultural lands protected through development easements, farms with agricultural deed restriction sold through the State Agricultural Development Committee, and land enrolled in eight-year preservation programs. The FPP data include attributes on the program under which the lands were enrolled and the duration of the protection program. I included these FPP lands in my analysis for two reasons: 1) to identify private lands protected through state agricultural programs which might have been excluded from other databases, and 2) to insure that patches of forest on preserved agriculture lands were included in the event they are contiguous with other protected forest regions. Inclusion of the eight-year FPP lands means that caution should be exercised in using this database. Some FPP lands may drop out of conservation status through time. Because my study focuses on protected forest land, I was not overly concerned with the potential later removal of some protected agricultural land and their associated forest. These forest patches would likely be small and exert little influence on my results if their conservation status changed.

The NJSCF data (NJDEP, 2011) was updated in 2014 and includes most, but may not include all, the purchases made by the NJ State Green Acres program since its inception in 1961. Although a full accounting of all Green Acres transactions is available as a spreadsheet of the block/lot number assigned at the time of acquisition, block/lot numbers have changed or undergone consolidations through time. The standardized New Jersey statewide GIS tax parcel dataset (known as “MOD IV”) has only been available since 2010 for mapping these Green Acres purchases. Matching every Green Acres spreadsheet transaction to the updated MOD IV block/lot requires archival investigation beyond the current resources of the Green Acres office (John Thomas pers. comm). Thus, NJSCF provides the best publically available spatial data on acquisitions from the Green Acres Program.

The compilation of these separate databases into a single relational database served two purposes. First, it brought together all publically available data on Protected Areas in the NJ Highlands into a single spatial dataset. Secondly, it improved the overall quality of the spatial dataset by integrating all available attributes on land ownership, acquisition dates, and resource management practices for PAs.

Aligning Protected Areas to Tax Parcels

I standardized all PA boundaries identified within the four GIS datasets to the tax parcel boundaries of the 2010 MOD IV tax parcel dataset. This required two steps. First, I created a spatial layer with centroids for each polygon in the four PA datasets and used it to select the MOD IV tax parcels containing that centroid. This process identified the majority of protected tax parcels. The HCPL dataset in particular was quite comprehensive and identified the majority of protected parcels using boundaries well aligned to MOD IV parcels boundaries. However, since manual inspection of the final data output revealed both errors of omission and commission, as a second step I overlaid each of the four GIS datasets onto the MOD IV data layer in which the centroid-based selections were visible. I manually selected the missing tax parcels, or removed incorrectly selected parcels. This PA dataset standardized to MOD IV and incorporating all my manual updates will be referred to as "PATX". The final version of the PATX dataset included protected lands with highly diverse land cover (e.g. recreation fields, farmlands, and wetlands). Portions of individual tax parcels which are counted as 'protected' also include infrastructure such as parking lots, buildings, and roads. This infrastructure exists to support recreational activities (Foresta 1981).

The MOD IV dataset is formatted using NAD 1983, NJ State Plane projection. This is the same datum and projection used for the HCPL, the FPP, and the NJSCF dataset. This common projection means that boundaries from the PA polygons in these three datasets conformed well to the tax parcel boundaries in the MOD IV dataset. PADUS, however, differs and therefore was subject to boundary misalignments when overlaid on the MOD IV layer. Since most PA polygons in the PADUS dataset were duplicated in my other datasets, the discrepancy in projection for this single dataset was not overly problematic. Because most PA datasets were well aligned with MOD IV, building the spatial dataset often only required the removal of polygon 'slivers' falling just outside the tax boundaries. These slivers are a common nuisance in spatial datasets and did not require any special decision rules to determine if a tax polygon should be considered protected.

However, in cases where one or more of the four PA datasets identified only partial acquisition of a tax parcel, I did have to generate decision rules. I assumed that a parcel subdivision occurred subsequent to the available MOD IV records. In these cases, I erased the unprotected portion of the tax parcel from the PATX dataset so as to avoid overestimating the total protected area. If there was disagreement among PA databases regarding the total preserved amount of a tax parcel, I always included the larger area. In other words, I included the maximum amount of information from all PA datasets, but I never represented individual tax parcels as fully protected unless I had the evidence from one of the databases to substantiate it. In this way, I was both inclusive of all available data, but was cautious not to over-represent the areal coverage of protection.

The PATX dataset I developed contained 10,401 protected tax parcels totaling 1274.4 km^2. In most cases, (10,279 tax parcels) visual inspection of the overlays showed obvious conformity between the polygon boundaries from the four original PA datasets and the tax parcel boundaries in the MOD IV dataset. In these cases, as explained above, spatially standardizing the PA polygons to the MOD IV data primarily required the removal of polygon slivers. The cases where one or more of the four PA datasets identified only partial acquisition of a MOD IV tax parcel were relatively rare. Within 122 tax parcels, I erased only 7.24 km^2 in total from the PATX dataset because I could not verify protection for the entire parcel. After making these corrections, the 10,401 protected tax parcels covered 36.6% of the ~3477 km^2 Highlands region.

Standardizing the four GIS datasets to MOD IV was necessary to clarify legal boundaries of PAs because land transactions (and therefore ownership and protected status) follow tax parcel boundaries. Standardizing PAs to the MOD IV parcel boundaries also preserved features such as roads and right-of-ways (ROWs) as sources of fragmentation within PAs. These were necessary to retain because these portions of the landscape are unavailable for protection and therefore create breaks in otherwise contiguous protected habitat. Consequently, in my analysis I did not count a state park with paved access roads throughout as a single large Protected Area, but rather as many smaller individual parts which I call "PA units". These were defined by both hard boundaries (e.g. roads) and soft boundaries (e.g. ownership/tax parcel boundaries). A detailed description and illustration of the process for identifying PA units follows.

Beyond serving several immediate needs for my analysis, the effort applied to aligning PA dataset to MOD IV boundaries also enables me to link protected parcels to their individual (and current) block/lot numbers. This creates the framework for relating the dataset to human social and economic data through tax records, and enhances the functionality of the data for future studies. One of the challenges in managing ecological systems within urbanizing areas is integrating data from both ecological and human landscape to understand how the systems feeds back and affect one another. Although this work focuses primarily on spatial patterns in forests, the dataset was structured to be dynamic and to scale up and outward to other datasets on land use, thereby making it a foundation for future studies on PAs.

Identification of Protected Area units and Protected Forest patches

Using a single township as an example, Figure 2.4 provides a visualization of how I derived two units of analysis, "PA units" and upland "PF patches", using the PATX data layer and the 2012 land use/land cover data. I defined PA units by dissolving all shared boundaries among any protected tax parcels in the PATX dataset. Dissolving boundaries did not change the sum of the PA area; it only merged individual adjacent PA tax parcels into single larger polygons (Figure 2.4a-b). Consequently, my PA units were defined via spatial boundaries, rather than administrative ones. In the context of my analysis, PA ownership was of interest only to the extent that it allowed me to distinguish between protected and unprotected land. An individual PA unit may have one or many landowners within it, but the unit itself I define by having contiguous protected habitat.

I delimited upland PF patches by first intersecting PA units with the 2012 land use/land cover (lulc) dataset and selected only polygons attributed as "Forest" within the Level 1 land cover classification included in the dataset. The lulc dataset uses a modified Anderson lulc classification system which includes five general ("Level 1") land cover categories: Agriculture, Barren, Urban, Forest, Water, Wetlands (NJDEP 2002). This lulc data is a statewide GIS product available online with detailed metadata on the land cover classification system (NJDEP, 2015).

Most "Forest" polygons in the lulc dataset are comprised of smaller individual polygons assigned to one of many forest cover sub-types (Figure 2.4c) for which the minimum mapping unit was one acre (0.405 ha). I dissolved the shared boundaries among any Forest polygons within a PA unit, merging forest cover sub-type. In this way, I used both the spatial extent of forest habitat and the legal boundaries of protected properties to define PF patches (Figure 2.4d). Note that in Figure 2.4, all PA units and forest boundaries are artificially truncated at township borders for the purposes of this example. In the full analysis, the dissolved procedure allowed all PA unit and PF patch boundaries to extend across all township borders occurring within my study site.

I binned both PA units and PF patches and into nine size classes in units of hectares (ha). The size classes were 0-1, 1-5, 5-10, 10-25, 25-50, 50-100, 100-500, 500-1000, and 1000-5000. The spatial distribution, count, and summed area of all size classes of PA units and PF patches are shown in Figure 2.5a-b to illustrate relationships between the total area protected and forests within those areas. Within the maps for Chapter 3, I inserted the Interstate-80 Highway corridor as a reference line to subdivide the Highlands into a northern and southern section, so this geographic reference is also shown the figures. The location of I-80 facilitated discussing regional spatial patterns of forest protection in the results sections of my analytic chapters. My analytic chapters focus exclusively on the forested portions of PAs, which comprised about 62% of the total area protected. In PAs, two size classes held the highest total amount of land (forested or unforested): the 100-500 ha class and the 1000-5000 ha class. However in PFs, the greatest amount of land occurred in the 100-500 ha class, with double or more than the amount of any other class.

The process of organizing and preparing the database raised questions about why ~2200 PA units and ~5500 PF patches under five hectares apparently had protected status given that there were small and spatially isolated from other protected tracts of land. For PFs specifically, the high frequently of small PF patches may occur because land transactions occur along tax parcel boundaries and thus small habitat patches are protected only up to that boundary. For PAs, it was more difficult to hypothesize about why such small areas were protected. Even if the aspiration was to spatially grow a small PA (or PF) through additional land acquisition, starting by protecting a land parcel or habitat patch under five hectares would seem a risky and unlikely investment for most land conservation agencies. The likelihood of any small protected parcel having other parcels appended to it through time certainly also decreases if there are thousands of these "seeds" planted throughout the regions, all competing for growth in an atmosphere of limited funding.

Using digital orthophotos, a visual review of a few dozen of the PAs and PFs under five ha suggested that they might be small-area easements on owner occupied private land, or that they may exist to create access routes to larger PAs. (Some were just confounding as they appeared to be highly landscaped residential properties.) Physical access to conservation land is a common concern among agencies engaged in land acquisition. Because most land is privately owned, it is not enough for an agency to acquire conservation property, the agency must also acquire land allowing for access to that property, generally with a vehicle. The two smallest size classes occupy a potentially interesting role in PF networks. If they are access routes, they may play an important role in the landscape by serving as small refugia between larger reserves. However the data for these classes would require additional filtering, validation, and a more targeted analysis to properly represent their role (e.g. is the land actively managed or monitored, or just a large backyard that offers a land owner a tax break). Since habitat quality is often related to the size of a habitat patch (Humphrey et al. 2015), and the function and conservation management of PFs under 5 ha was uncertain, I chose to focus my analytic chapters on the larger size classes. With these, I had higher confidence that I was representing areas intended to support biological conservation.

By using only the Level 1 land cover classification, I excluded wetland forests from my analysis. These are a special sub-category of wetlands added to the Anderson categories for the NJ land cover dataset in order to accommodate the multiple habitats overlapping in wetlands (Hasse & Lathrop 2007). Although less abundant than upland forests throughout the Highlands, wetland forests are often contiguous with upland forests and create potential points of connectivity between patches of the two forest types (Figure 2.6a). If included in this analysis as “Forest” cover, wetland forests could therefore potentially expand the area of many individual Protected Forests (Figure 2.6b) and individual Forest Fragments (an analytic unit I introduced in Chapter 3). The implications of expanding my definition of “Forest” to include wetland forests are specific to each research question and are therefore discussed within each analytic chapter.

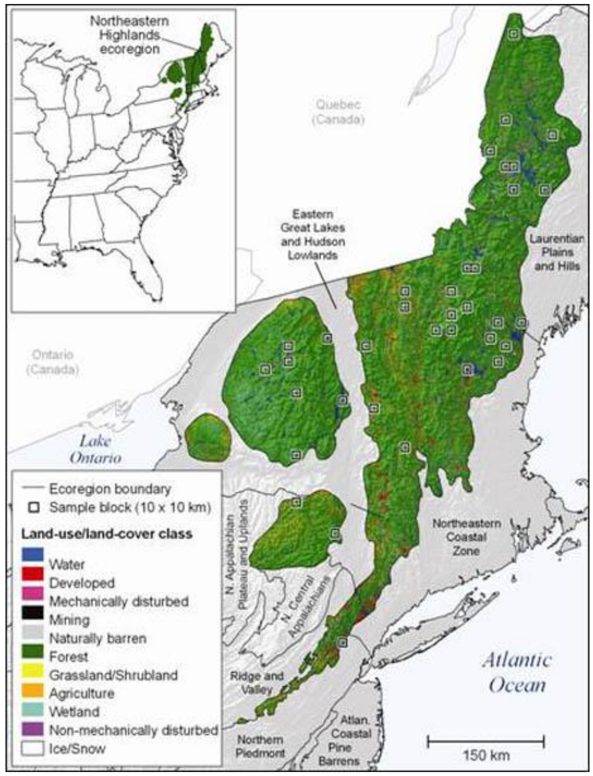


Figure 2.1 The Northeastern Highlands ecoregion as defined by the United States Geological Survey criteria for ecoregions (source: USGS, 2016b).

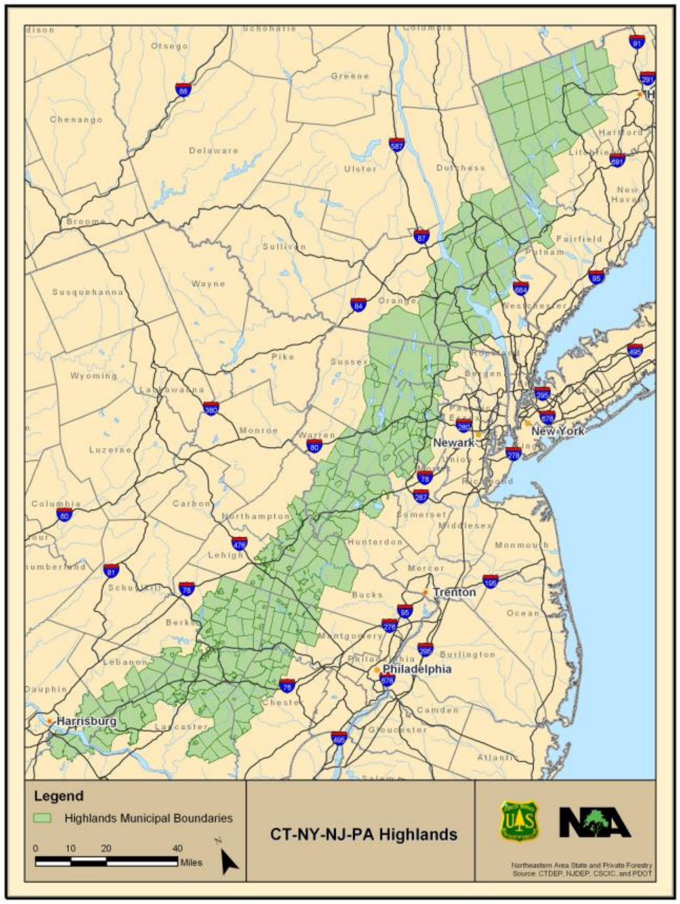


Figure 2.2 The subsection of the Northeastern Highlands targeted in the Highlands Conservation Act of 2004 (source: USGS, 2016b).

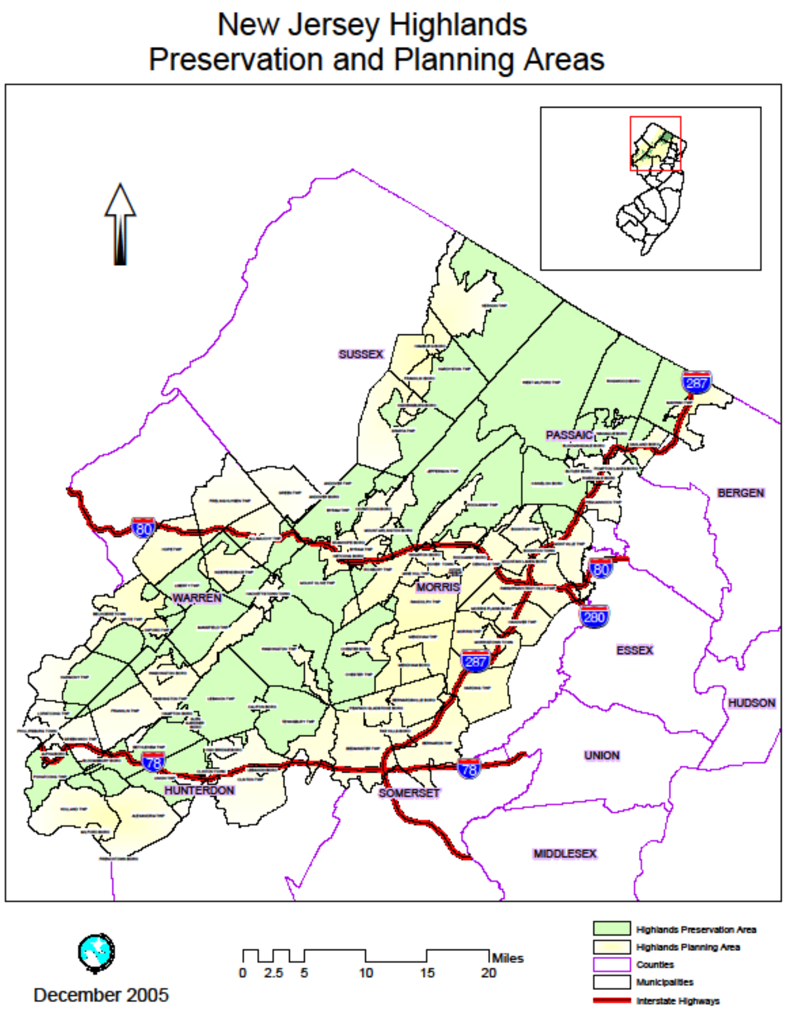


Figure 2.3 The NJ municipalities which fall into the Highlands Preservation (green) and Planning (yellow) Areas. The Preservation Area has stricter regulations than the Planning Area (source: (NJDEP, 2005).

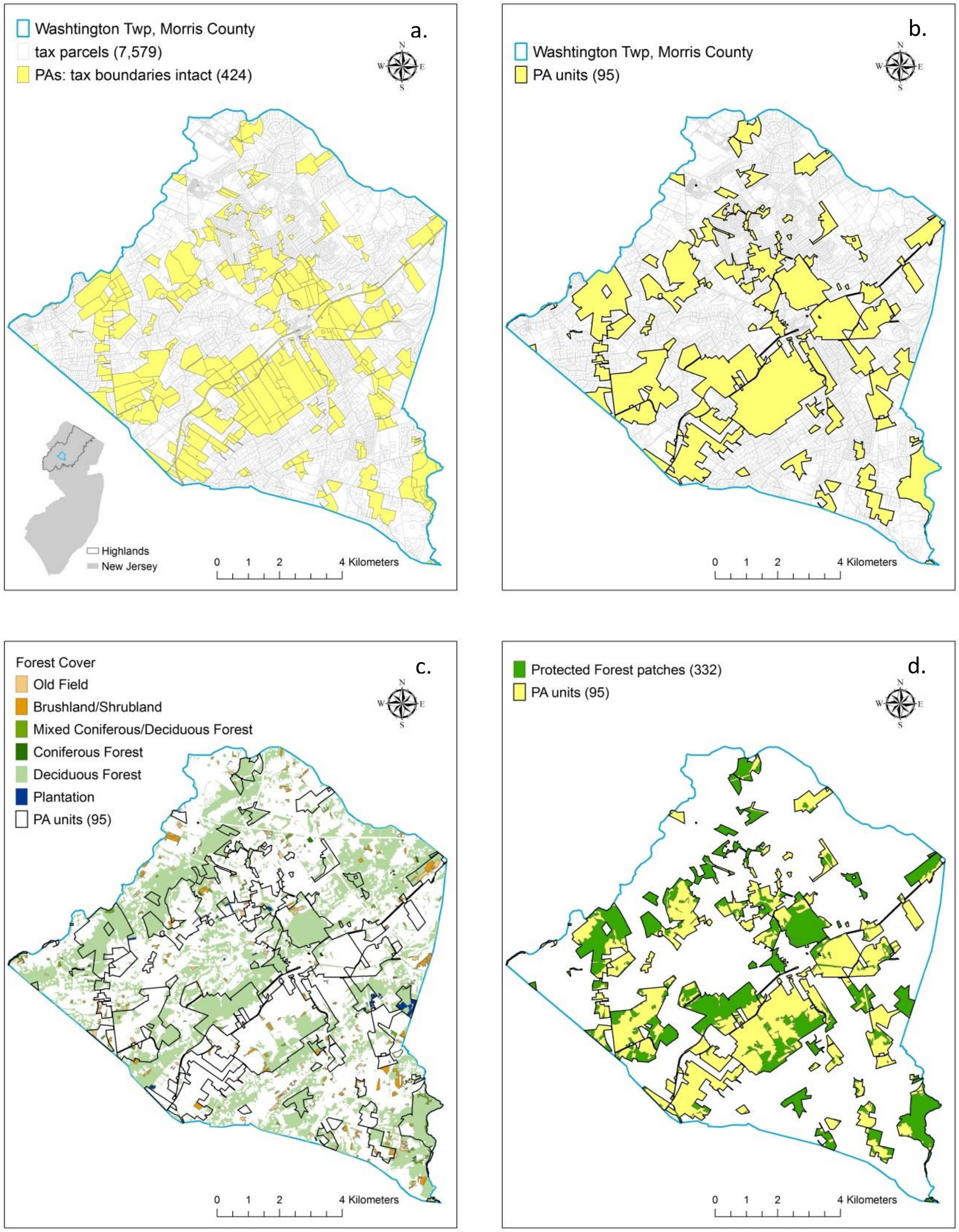
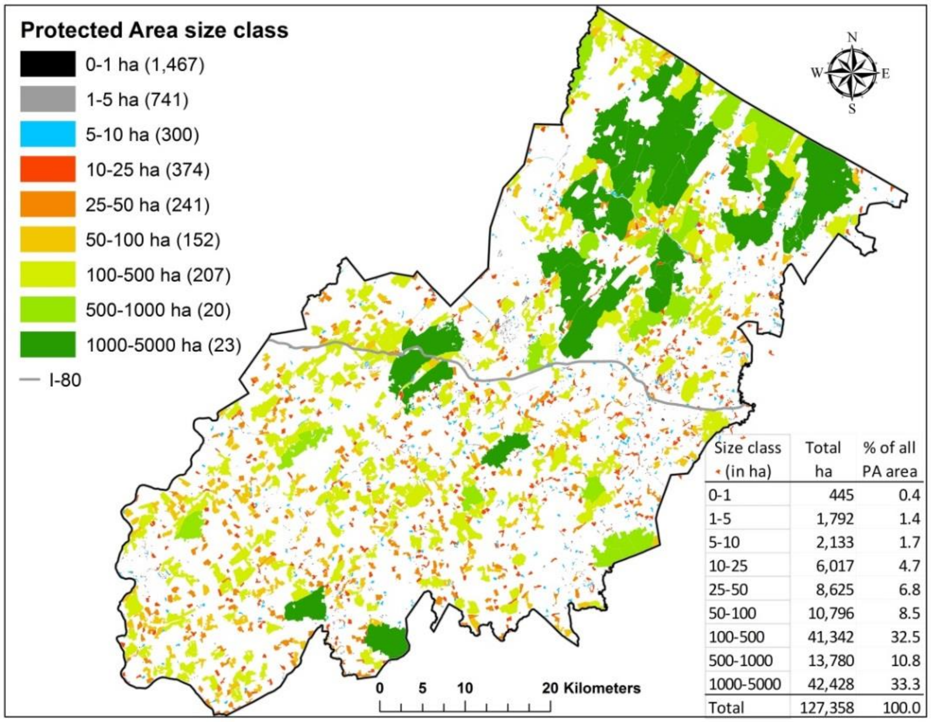
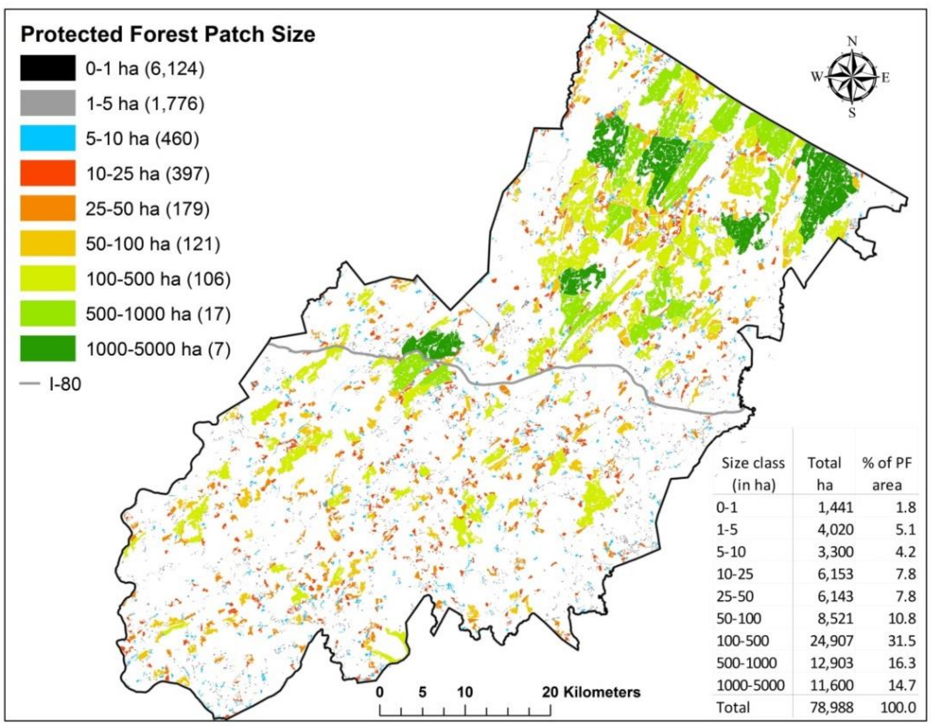


Figure 2.4a-d Using the forests in Washington Township, Morris County as an example, this figure illustrates the process for identifying (a) Protected Area tax parcels, (b) PA units, (c) forest types within PAs, and (d) the Protected Forest patches within PAs. Counts of polygons in each category are shown in parentheses within the legend.

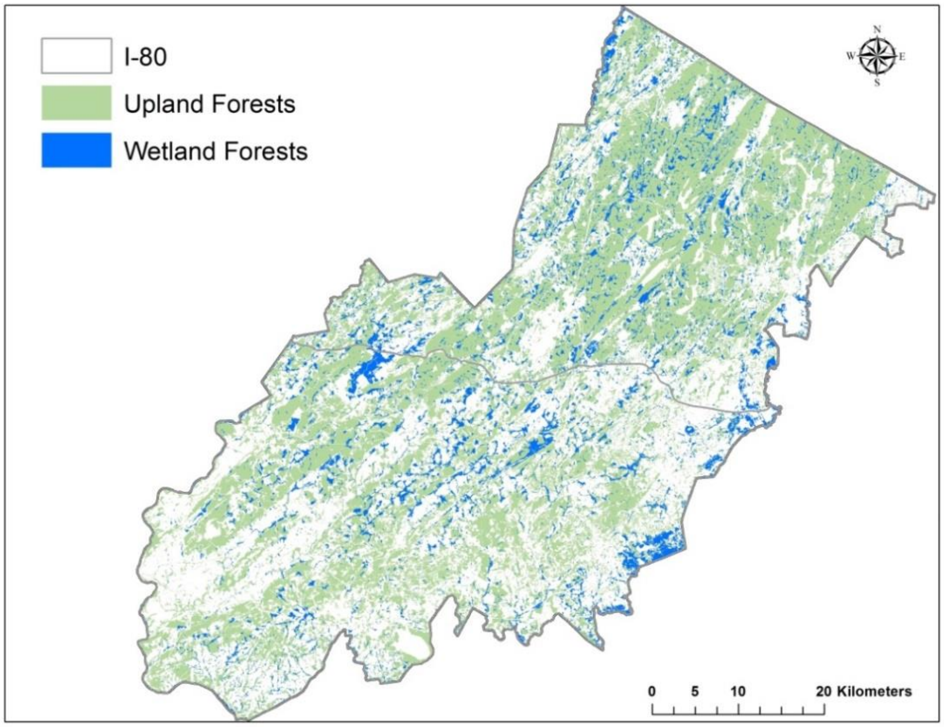


Size class (in ha)	Total ha	% of all PA area
0-1	445	0.4
1-5	1,792	1.4
5-10	2,133	1.7
10-25	6,017	4.7
25-50	8,625	6.8
50-100	10,796	8.5
100-500	41,342	32.5
500-1000	13,780	10.8
1000-5000	42,428	33.3
Total	127,358	100.0



Size class (in ha)	Total ha	% of PF area
0-1	1,441	1.8
1-5	4,020	5.1
5-10	3,300	4.2
10-25	6,153	7.8
25-50	6,143	7.8
50-100	8,521	10.8
100-500	24,907	31.5
500-1000	12,903	16.3
1000-5000	11,600	14.7
Total	78,988	100.0

Figure 2.5a-b The spatial location of PAs (top) and PFs (bottom) in each size class are mapped. The count for each size class is shown in parentheses. The inset has the summed area for each size class.



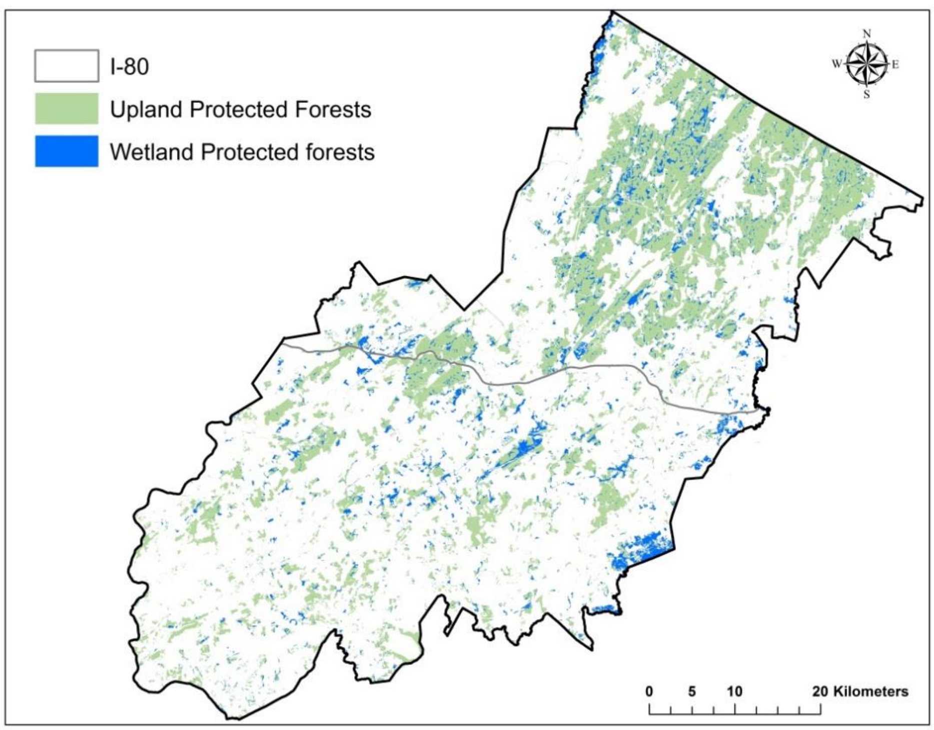


Figure 2.6a-b The spatial arrangement of upland forests relative to wetland forests (a). The spatial arrangement of upland forests in legally protected parcels, relative to wetland forests in legally protected parcels (b).

Chapter 3: Connected, but not protected: Forest parcelization in the NJ Highlands and its implications for Protected Forests

Introduction

Over the past few decades, Protected Areas (PAs) have become the natural resource conservation tool of choice for many agencies worldwide in response to the increased human demands on the environment (Gaston et al. 2008). PAs exist to meet a variety of resource conservation goals such as timber production and the protection of cultural heritage sites, water quality, and biodiversity (Locke & Dearden 2005; Dudley 2008). For Protected Areas which form biodiversity reserve networks, the main goals in selecting parcels for inclusion are to 1) represent and 2) maintain biodiversity long-term (Margules 2000). Meeting these goals depends increasingly upon the spatial arrangement of PAs within human land use systems, and the stability of that arrangement through time (Cummings 2015).

Land acquisition agencies of all sizes and capacities are often obliged to incorporate biodiversity protection into their land selection in the absence of perfect knowledge about the species and landscape processes that threaten the long-term viability of species (Soule 1985; Cabeza & Moilanen 2001; Watson et al. 2016). In the absence of such knowledge, for decades general spatial principles have often been used as proxies to guide decisions. Two such principles are 1) protect the largest available habitat patches (rather than small ones), and 2) protect entire habitats patches (rather than fragmenting them) (Primack, 2008; Shafer, 1997). However, setting aside large contiguous habitat patches is a challenge in the U.S. because of the way land conservation policies are structured. Land condemnation for conservation, although technically permissible by law, is rarely invoked in deference to the rights of private landowners (Foresta 1981; Fairfax et al. 2005). Protected Area selection is therefore often based on the 'availability' of land, (i.e. the landowner's willingness to sell, donate, or otherwise commit the land to conservation management), and the capacity of conservation agencies to respond to these opportunities (Fairfax et al. 2005; Knight & Cowling 2007; Knight et al. 2011a).

This land acquisition practice has two consequences that yield similar spatial outcomes. The first consequence is that most land transactions occur along tax property (land use) boundaries rather than ecological (land cover) boundaries. Thus, many Protected Areas may only be smaller portions of larger contiguous habitats. The second consequence is that this practice favors opportunistic acquisition over systematic PA network design (Foresta 1981; Margules 2000; Mcdonald 2009; Knight et al. 2011a). Therefore, if willing sellers do not own adjacent parcels of desirable conservation land, this may also result in PAs occupying small portions of larger contiguous habitats. Both conditions make it difficult to assemble PA networks with the large and fully protected habitat patches preferable for biodiversity conservation in a dynamic anthropogenic landscape.

Forest fragmentation versus forest parcelization

In the context of conservation forest fragmentation, rather than forest parcelization, is the landscape process which has been more widely studied (Debinski & Holt 2000; Haila 2002; Olff & Ritchie 2002; Ewers & Didham 2006; Prugh et al. 2008). The term "fragmentation" has been inconsistently applied to explore various aspects of habitat loss, change in habitat configuration, and relationships between a habitat patch and the surrounding landscape composition (Fahrig 2003). However, one common denominator in all fragmentation studies is the "breaking apart of habitat, independent of habitat loss" (Fahrig 2003). Forman's work on land mosaics (1995) describes the diversity of landscape factors which can break the continuity of land cover and thus shape the nature of fragmentation. Fragmentation may have human or non-human origins (e.g. forest habitat is fragmented by both agriculture and natural water bodies). The land cover changes associated with habitat fragmentation may occur abruptly over short distances, creating distinct ("hard") edges. Transitions may also occur over larger distances, creating more subtle ("soft") edges (Forman 1995).

"Parcelization" of forests differs from fragmentation of forests. Whereas fragmentation describes a discontinuity of land cover, parcelization describes a discontinuity of ownership occurring when larger forested tracts are subdivided into smaller parcels and transferred to different individuals (Germain et al. 2006). Any given forest fragment may be owned and therefore managed by multiple individuals or agencies, each controlling a distinct spatial piece. Management among owners within a forest fragment need not be cooperative and may even be contentious. Therefore parcelization creates the potential for decreased habitat quality through time within a contiguous forest tract (Knoot et al. 2009, 2010; Schaich & Plieninger 2013)

Causes and consequences of parcelization

A variety of social and economic conditions have been linked to the process of forest parcelization. Transitions in U.S. corporate forestry markets and practices have restructured ownership patterns, expanding the role of smallholders in forestry management (Bliss et al. 2010). At smaller scales, rising land values and increased property tax burdens have been linked to the subdivision of non-industrial private forests (NIPF) (Meehmood & Zhang 2001; D'Amato et al. 2010; Mundell et al. 2010; Stone & Tyrrell 2012), although that link is disputed in at least one study (Kilgore 2014). Gruver et al.'s (2017) work on NIPF legacy decision-making highlights the complexity of factors that owner encounter outside economic variables. They argue that factors such as family relationships and access to planning information interact with economics to drive choices about subdividing forests.

Within forestry literature, parcelization of contiguous forests has received attention due to its implications for overall forest productivity and health (Gustafson & Loehle 2006; Stone & Tyrrell 2012). At a minimum, parcelization complicates the management of contiguous forests. Having multiple stakeholders within a forest diversifies the socio-economic pool of owners and, in turn, influences the type of management and use that different owners support (Brenner et al. 2013). For example, throughout the northeastern U.S. where white-tailed deer are known to have significant impacts on forest composition and regeneration (Côté et al. 2004; Williams & Ward 2006; Baiser et al. 2008; Urbanek et al. 2012), parcelization complicates herd management. The increase of new landowners at the urban-wildlife interface introduces diverse opinions about human safety, hunting access, hunting capacity, and the role of wildlife, weakening the use of hunting as a deer management tool (Campa et al. 2011).

Forest parcelization can also increase spatial heterogeneity across forest cover. Ownership boundaries often show spatial correlation with different land cover types (Crow et al. 1999; Croissant 2004). The spatial connectivity of forests may be altered via the owner's management of wildlife (Sandström et al. 2013), timber harvest (Crow et al. 1999; Schaich & Plieninger 2013) and fuel loads (Busby et al. 2012). Ownership also gives individuals the right to permanently deforest their property. Therefore, with every additional landowner or investor within a contiguous forest comes the additional possibility that part of that forest will be developed if forestland is costly or difficult to manage (Phelps & Hoppe 2002; Butler & Ma 2011). For example, Mundell et al. (2010)

found that on 68% of forest parcels subdivided during a given year, development followed within seven years. Parcelization, therefore, creates many conditions which can disrupt habitat connectivity and may favor future forest fragmentation (Germain et al. 2006; Caron et al. 2012; Stone & Tyrrell 2012).

Parcelization and biodiversity

Forest parcelization is a somewhat underrepresented topic in the literature on biodiversity conservation and management, but it should not be. Just as in forestry, it is an important consideration because it complicates management of Protected Areas. Protected Forest parcels (PFs) embedded within larger forest fragments do not function as independent entities. Regions where protected and unprotected parcels conjoin often incubate resource disputes due to conflicting practices of owner consumption and production (Harrison et al. 2004; Ostrom & Nagendra 2006; Rudel et al. 2009; DeFries et al. 2010).

Just as in forestry, parcelization is also an important consideration for biodiversity because it can increase forest spatial heterogeneity and compromise overall forest habitat quality. Gustafson et al.'s work (2007) argues that the sustainability of any forest species' population does not occur at the scale of the individual landowner. Rather, population viability is an aggregate function of management by multiple landowners at regional scales, generating a dynamic mosaic of forest types, stand structures, and age distributions. Individual landowner actions can aggregate across a habitat or region to shape the landscape mosaic and its ability to sustain forest communities and their ecological processes (Gustafson & Loehle 2006, 2008; Gustafson et al. 2007). The viability of natural resources and wildlife populations within any legally protected forestland is therefore subject to change if other landowners in the larger contiguous forest change their management practices. Although spatial heterogeneity may benefit some species or aspects of natural resource conservation, the ability to sustain and protect others will be compromised (e.g. see Di Ionno, 2016 regarding the timbering of Sparta Mountain, NJ). It is therefore necessary to understand how contiguous forest is parcelized into unprotected and legally protected tracts of land.

Parcelization in northern U.S. forests

Much of the recently compiled data on parcelization for NIPF owners of the northern U.S. comes from two sources (Butler 2008; Butler & Ma 2011) which analyze U.S. Forest Service data. The work confirms a previously untested assumption that there have been regional increases in parcelization among this class of landowner. At the beginning of this century, in the northern U.S. there were 4.7 million family forest owners who collectively held 55% of all forestland (Butler 2008). From 1993-2006 holdings decreased from an average of 25 ha to 20 ha (Butler & Ma 2011). The smallest average holdings for family forests (i.e. those holdings less than 10 acres) were found in the most densely populated states along the northeastern coast (NJ, MD, MA, CT, DE, RI). Accompanying this decrease in area was a demographic shift in the average owner. On average family forest owners in 2006 were older, more educated, and had a higher income than those in 1993, and there was a decrease in the percentage of farmers (Butler & Ma 2011). These demographic factors are relevant because they can influence the size and location of parcels which become available for incorporation into forest conservation networks. Smaller individual land holdings for each owner may result in highly piecemeal protection in the largest remaining forest tracts.

Objectives

In eastern U.S. forests, private ownership pre-dated the interest and legal pathways for conservation land acquisition conservation, so most Protected Forest networks have been assembled piecemeal via donation or buying back individual parcel (Fairfax et al. 2005). Consequently, patterns of partial forest protection are likely to be particularly evident there, especially in the most densely populated states where parcelization has resulted in the largest percent decrease in average size of holdings (Butler 2008; Butler & Ma 2011). The objective of this study is to quantify the patterns of protected versus unprotected forest within the forest fragments of one eastern U.S. urbanizing region. Using the NJ Highlands as a case study, I examined the size and distribution of upland Protected Forests patches (PFs), and the extent to which they are embedded with larger analytic units which I call upland Forest Fragments (FFs) (described in the next section). This study addresses a gap in conservation research by quantifying forest parcelization through the lens of Protected Area networks. The frequency and location of partial protection in forest fragments has implications for sustaining biodiversity in existing PAs because it may facilitate future forest fragmentation.

Methods

I reduced my study of parcelization within the Highlands forests to a binary of protected versus unprotected regions as a way to represent different management regimes in a contiguous forest tract. The process of land acquisition is a complex one. Conservation lands are often acquired through multi-agency partnerships (Foresta 1981; Fairfax et al. 2005), and can be transferred to the management or ownership of other agencies while retaining protected status. Assigning ownership and doing a more detailed analysis based on owner classifications (e.g public versus private versus NGO) or management practices was both legally and temporally complex and fell beyond the scope of this study. The binary of protected versus unprotected does not suggest that all PF owners have uniform management practices, or that unprotected land is always managed differently than protected land. However, it indexes the future land cover in forested tax parcels as either largely predictable (protected), or unpredictable (unprotected). This provides a understanding of habitat vulnerability to loss within a contiguous forest.

Defining forest fragments for this study

Drawing on the definitions provided by Fahrig (2003) and Forman (1995), I use the term "forest fragmentation" or simply "fragmentation" throughout this chapter when referencing the spatial discontinuity of forests resulting from land cover change. In my use of this term, I imply no minimum amount of forest loss, no specific source of forest loss (i.e. anthropogenic versus natural events), and no specific edge characteristics (e.g. hard or soft). All forest polygons were pre-defined in the 2012 land use/land cover (lulc) dataset I used. This is a publically available dataset produced by the NJ state government (NJDEP et al. 2015) and the procedure for defining the boundaries between "Forest" polygons and other land cover types is provided within that metadata.

To delineate upland Forest Fragments as units of analysis for this study, I used a GIS process similar to the one for identifying Protected Forests (previously described in Chapter 2). First, the adjacent boundaries for all tax parcels within the MOD IV tax layer were dissolved (refer to Chapter 2 for a full description of the MOD IV dataset and its source). Because features such as roads and right-of-ways (ROW) are excluded from much of MOD IV, the dissolve procedure preserved these features as sources of fragmentation in land cover. Although some ROWs, (e.g. powerlines) do support forest cover, that cover is maintained in early successional stages (Russell et al. 2005; Bulluck &

Buehler 2006) creating linear disturbance corridors within forest cover (Forman 1995). Because the boundaries and successional stages of the forest cover in ROWs are somewhat fixed spatially and temporally, and because it is unlikely these areas would become available for protection unless utilities services change, I chose to include all ROWs as non-forest features which help to define forest fragment boundaries.

Second, I used the merged tax polygons (minus roads and ROWs) to clip the 2012 NJ land use land cover data layer. I selected only polygons with the Level 1 “Forest” attribute from the clipped lulc dataset (see Chapter 2 for a description of the lulc data). I dissolved the boundaries among forest sub-type polygons to produce a GIS layer of all individual upland FFs. To each FF polygon, I assigned a unique identifier used to link each to the one or many PFs embedded within them. PF patches were spatially defined by two simultaneous conditions (land cover = forest *and* land use = protected, Chapter 2). Using the terminology ‘PF patch’, or simply ‘PF’ throughout this text distinguishes protection as a form of *land use* parcelization which was layered upon the *land cover* fragmentation accounted for in the units of FFs.

I binned FFs using the same categories used for PFs (0-1 ha, 1-5, 5-10, 10-25, 25-50, 50-100, 100-500, 500-1000, 1000-5000), calculated summary data for all FF size classes, and mapped their extent. I aggregated the area data for FF size classes and PF size classes greater than five ha and compared the two data distributions. Within the maps, I included the Interstate-80 (I-80) highway corridor to subdivide the Highlands into a northern and southern section in order to facilitate discussing regional spatial patterns of forest protection.

Spatial relationships between Forest Fragments and Protected Forest patches within them

I spatially intersected the layer of PF patches with my layer of Forest Fragments (Figures 3.1a-c) in order to examine spatial relationship between the two. Conserved tax parcels determined the invisible or non-physical boundaries of Protected Forests. One or many individual tax parcels (shown by the grey lines in Figure 3.1a) may compose a contiguous PF patch (the green region of Figure 3.1c). The tax parcel boundaries determined whether PFs within a single forest fragment were spatially separated from one another (Figures 3.1b-c). As an example, the forest fragment in Figure 3.1c has three spatially disconnected PFs of differing sizes within the larger fragment.

I linked PFs to the FFs in which they were embedded using a combination of tabular identification keys and the spatial join tool in ArcGIS 10.2 (ESRI, 2014). The zonal statistics tool was used to calculate aggregate values for percent protection within FF size classes. Ideally, every FF would have full protection, but this ideal condition does not frequently occur in practice and the deviations are therefore informative. For example, if the 1000-5000 ha FF class had only PFs of the same size within it, it reduces the risk of future fragmentation. Conversely, if the 1000-5000 ha FF class had only PFs less than 100 ha within it, it potentially elevates the risk. (The risk changes because there are a lot of unprotected interstitial forested spaces between PFs which could be deforested.)

The zonal statistics tool was also used to calculate and map the percent protection for individual FFs. To map the percent of forest protected within individual FF, I summed PF area for each FF, binned the results into five categories (0%, 1--25%, 25-50%, 50-75%, and 75-100%) and passed that information back to each FF as an attribute. Individual FFs were colored according to the percent of forest protected. Because conservation goals in the Highlands include maximizing the amount of interior forest lands protected, rather than forest edge (Phelps & Hoppe 2002), I focused on mapping patterns of protection within individual FFs of the three largest FF size classes (100-500 ha, 500-1000 ha, and 1000-5000 ha). Because of their size, these three largest size classes also have the highest potential to have many small, spatially distinct PFs within them. Thus, identifying patterns of protection were particularly relevant for these groups.

Results

Distributions of FFs and PFs

My analysis identified 18,832 spatially distinct upland Forest Fragments totaling ~1550 km^2 (this sum includes protected and unprotected forests). The spatial distribution of all sizes across the study site is shown in Figure 3.2, along with summary data for each class. Forest fragments greater than 500 ha were concentrated north of I-80. Fragments under ten ha were not visibly concentrated in either the northern or southern regions; they occurred throughout most of the interstitial space between larger-sized fragments.

The highest FF counts generally occurred in the smallest size classes and decreased as class size increased. One exception to this pattern was the 100-500 ha class, which had a slightly higher count of fragments than the size class below it (264 vs 243). The total hectares of forest represented by each fragment size class did not show strong patterns related to size class. Collectively, the three largest size classes held the highest amounts of forested land - just over 60% of the total forest extent in the region. However, the 100-500 ha size class contained about twice as much forest cover as the two size classes above it. This highlights the prominent role upland forest fragments 100-500 ha occupy within the NJ portion of the greater Highlands ecoregion.

Figure 3.3 shows the spatial distribution of protected versus unprotected forest. Forests and protected forestland was not evenly distributed across the study site. Using total area as an index, more land was protected in the northern end of the Highlands. Comparatively less total forest area was protected south of I-80. Areas of contiguous protected forests were also generally smaller and more numerous in the southern end, showing that the remaining forests in the south had more landscape features fragmenting them.

The upland PFs in Figure 3.3 are quantified and graphed in Figure 3.4a-b to compare the total area for PFs (3.4a), versus that of FFs (3.4b) for all classes above five hectares. The two graphs had somewhat similar distributions, with the maximum summed area occurring in the 100-500 ha size class. The two largest FF size classes combined had less total area than the 100-500 ha size class (42,004 ha vs 52,867 ha respectively). This was not true for the PFs, however. For the PFs, the total area in the two largest size classes nearly equaled the 100-500 ha size class (24,502 ha versus 24,906, respectively). The results reinforce the pattern noted earlier. Not only are the 100-500 FFs significant in the Highlands landscape because they hold the highest amount of forest, they are also significant units from a conservation perspective because they contain the most total protected forest land.

The two graphs in Figure 3.4 show two distinct data distributions and cannot be overlaid because not all PFs in one size class necessarily occurred within the corresponding FF size class. Since one goal of this analysis is to determine how frequently smaller PFs are embedded within larger FFs, this relationship is illustrated and quantified for each size class within Figure 3.5. The top figure shows the aggregated amount of land in each FF class and the relative percent protected of that aggregated area. There was a clear trend. The percent protection increased across size classes, with the largest FFs having the highest total percentage of land under protection. Eighty percent or more of the forest area in the two largest size classes was protected. The next largest size class (100-500 ha) had just over 40% of the land protected. Although this size class had the largest total amount of forested land and the greatest total amount protected (Figure 3.4), it did not have the highest percent protection. By this metric, conservation acquisition, either by design or opportunity, has favored forest protection within the largest FFsize classes.

Figure 3.6a-b maps the spatial distribution of forest fragments in the largest three size classes, and their individual percent of protection (with percentages binned into five classes). The largest FFs occurred disproportionately in the northern portion of the Highlands (n=36 versus n=9 in the south, Figure 3.6a). Smaller PFs (100-500 ha) occurred disproportionately in the south (n=98 vs n=166 in the north, Figure 3.6b). The north also had the highest proportion of large FFs with greater than 75% protection (29/36 or 81% in the north, versus only 4/9 or 44% in the south). This protection pattern was similar within the 100-500 ha size class. In this size class, 43% (42/98) had greater than 75% protection in the north. In contrast, only 11% (18/166) of those in the south had this level of protection. The majority of 100-500 ha FFs in the southern Highlands (67%) had less than 50% protection (indicated in black, red, and yellow on the map).

The boundaries of FFs and PFs represented in all figures are based on the criteria described in Chapter 2 and therefore resulted both from natural land cover like rives and wetlands and anthropogenic features like roads, ROWs, and agriculture. Wetland forests were not included in my delineation of PFs and FFs. The implications of excluding this land cover type are addressed in the discussion section of this chapter.

Discussion

The primary goal of this analysis was to consider how parcelization of existing forest fragments into protected and unprotected geographic regions might affect the long-term sustainability of the biodiversity within them. I argued that in the absence of perfect knowledge about present and future processes impacting biodiversity, the acquisition of large FFs (rather than small ones), and the full protection of individual FF (rather than partial protection) are spatial patterns favored in land conservation reserve design. Mapping these spatial patterns provides a rough index of vulnerability within individual Protected Forests, and how that risk is distributed across FF size classes and space. For FFs of any size, less protection means the embedded PFs have less certainty about the future of the contiguous forest cover on which PF resource sustainability depends. Within the largest remaining FFs, assessing this risk is of particular interest because maximizing the amount of protected interior forest is a regional conservation priority.

Despite the fact that acquisition opportunities for land were governed by the constraint of willing sellers, having greater than 75% protected forest was the dominant spatial pattern for the two largest size classes (Figure 3.6a). Further, more than half the embedded PF area occurred as PF patches equal to or just under the corresponding FF class size (Figure 3.5). High percentages of protection in the largest remaining forest fragment are consistent with preferred spatial patterns of network design. This suggests that NJ has made significant steps toward insuring the sustainability of regional biodiversity. However, since the largest fragments occurred disproportionately in the northern Highlands, this pattern also has some negative implications. Conservation acquisition, either by design or opportunity, has favored forest protection in one geographic region.. While this has positive implications for protecting resources in the northern Highlands, the natural resources in the southern portion of the NJ ecoregion have received less attention.

There are few large FFs in the south from the two largest classes. Forest Fragments 100-500 ha having less than 75% protection dominate this geographic region (Figure 3.6b). In aggregate, this size class had just under 50% of its lands protected (Figure 3.5). This size class could potentially play an important role in supporting resource sustainability across the regional Highlands network, but overall has less land protected at present. The existing PF land within this size class is threfore highly vulnerable to experiencing future habitat change because so much of it's contiguous forest is unprotected.

The high amount of land regionally contributed by the 100-500 ha size class, but the low percent of protection within it has two policy implications. First, this an important patch size within NJ to target for forest fragmentation studies. Understanding the feedbacks between habitat size and human land alterations around patches this size will be central to designing forest management practices and acquisition policies with wide application. Because there is so much land in this size class and it interface with many dimensions of human land use, it also offers high potential returns on any investment into management-oriented fragmentation studies.

Secondly, since these FFs represent the remaining upland forests throughout the south, they should be prioritized for acquisition in an effort to sustain the habitat quality across the greater Highlands ecoregion. The current PA network configuration creates a vacuum of forest protection directly in the center of the greater Highlands ecoregion (Chapter 1). This could have important implication in the context of a change in species' range, migration, or dispersal patterns (Opdam & Wascher 2004; Hannah et al. 2007;

Oliver et al. 2016; Poudyal et al. 2016; Titeux et al. 2016). Should the southern NJ Highlands PFs be restricted to their current spatial extent and additional fragmentation occur within them, this geographic region could effectively create an undesirable landscape filter for species which depend on forest interiors. Thus, even if large PF networks exist to the south in the Pennsylvania Highlands, and to the north in the NJ/NY Highlands, species' poorly adapted to use the smaller more fragmented PFs of the southern NJ Highlands may experience population reductions.

These results are instructive for many emerging PA networks in urbanizing regions. New Jersey has been a leader in open space protection at the state, county, and municipal level since the 1960s (Kotchen & Powers 2006; Heintzelman et al. 2013). Local ballot issues which have funded NJ open space acquisition are also being adopted across the U.S. (Lerner et al. 2007; Nelson et al. 2007; Szabo 2007). Nelson et al. (2007) suggests that such municipal-level initiatives could have major impacts on the spatial patterns of U.S. protected networks because conservation lands are purchased disproportionately in the locations which allocate funding. However, the process of PA network assemblage is winding down in the New Jersey Highlands as available funds dwindle and regions reach the limits of current zoning laws (Lathrop et al. 2007). Without doubt, there will be some expansion of PFs through additional purchases of currently unprotected land. Regulations like the Highlands Act of 2004, and policies governing development on steep slopes and wetlands (Rome 2001) also help insure that many unprotected portions of forests will remain as forest.

The present configuration provides a portal into future challenges PFs will face in sustaining biodiversity by virtue of their spatial arrangement. The patterns of partial forest protection documented throughout the southern Highlands in this study might be avoided in other states if municipal-level conservation activities are coordinated at a regional level and multiple conservation tools are incorporated into the process at earlier stages. New Jersey's Highlands Act mandates and facilitates regional planning, but most land acquisition in this study occurred before this legislation. The imprints of regional planning efforts are therefore emerging slowly, but may not be strong visible within these results. Other urban areas in less advanced stages of land acquisition would benefit from early implementation of regional planning legislation.

As land prices rise in a region, the outright purchase of conservation land becomes an exercise in diminishing returns (Ando et al. 1998; Vandegrift & Lahr 2011; Withey et al. 2012). Expanding and improving conservation efforts in other urbanizing regions across the U.S. may require recognizing the limits of acquisition as a conservation tool (Fairfax et al. 2005; Locke & Dearden 2005; McDonald & Boucher 2011) and investing in other approaches (Cowling et al. 2010; Knight et al. 2011b; Mora & Sale 2011; Cummings 2015).

The focus of this analysis has been upland forests. As noted in Chapter 2, many wetland forests are contiguous with upland forests and could be included to expand the definition and size of the FFs and PFs analyzed. As a consequence of appending wetland forest onto my FFs, many FF would increase in area and therefore become members in a larger size class. The addition of wetland forests would therefore decrease the number of Forest Fragments in class sizes under ten hectares and increases the numbers in size classes above ten ha (Appendix A). This has only a small effect on changing the relative percentage of total forest represented by each FF size class. However for PF patches, the relative percentages of the 100-500 and 1000-5000 ha size classes become more even. This has some potential implications.

If wetland forests are included, PFs 100-500 ha as a size class dominate less of the total forest area. This could potentially shift the message about prioritizing management and acquisition in this size class simply because the new data distribution mean there is yet more unprotected lands in the largest size class to acquire. However, given that forest protection is comparatively low in the southern Highlands and the largest FFs there are 100-500 ha, I would still argue for prioritizing this size class in acquisition policy and management studies.

Many legally protected parcels include some forested wetlands, but those which are not included still have had legal protection through the NJ Freshwater Wetlands Protection act of 1987 (N.J.S.A. 13:9B-1 et seq.). Given this, there is good reason to exclusively focus on protection and land change around upland forests. These are the forests with the greater total habitat area, but a less certain future as they can transition into either protection or development. Furthermore, treating wetland forests as contiguous with upland forest would not fundamentally alter the results concerning where protection occurs. Other results which have not been included here show that the bins used for both size class (Figure 3.5) and percent protected (Figure 3.6) were large enough to absorb the additional wetland forest data without shifting the results in the graphs and figures dramatically.

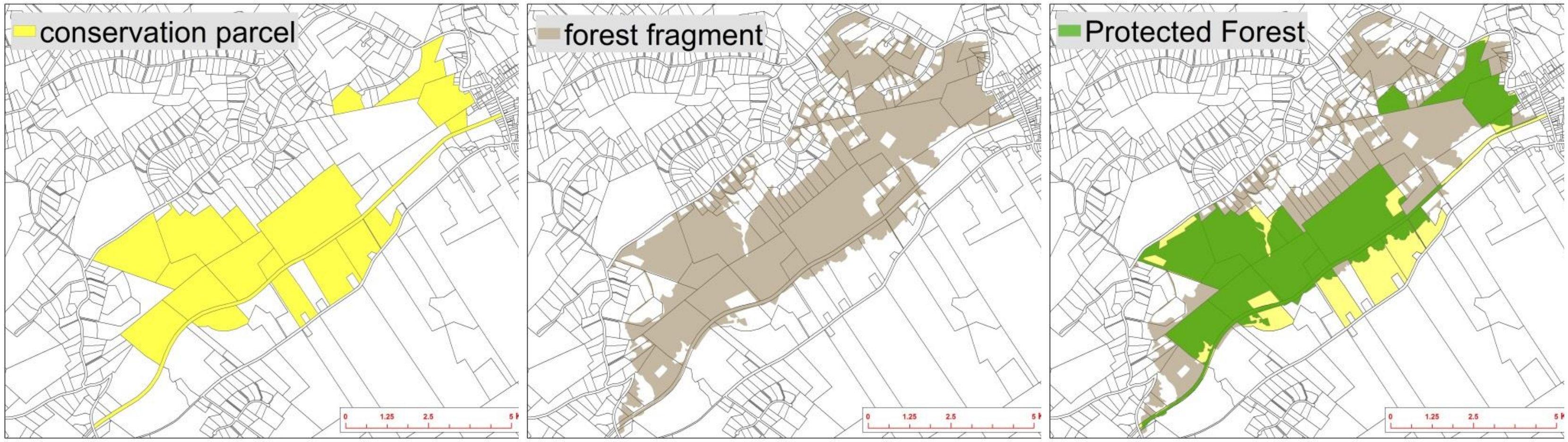
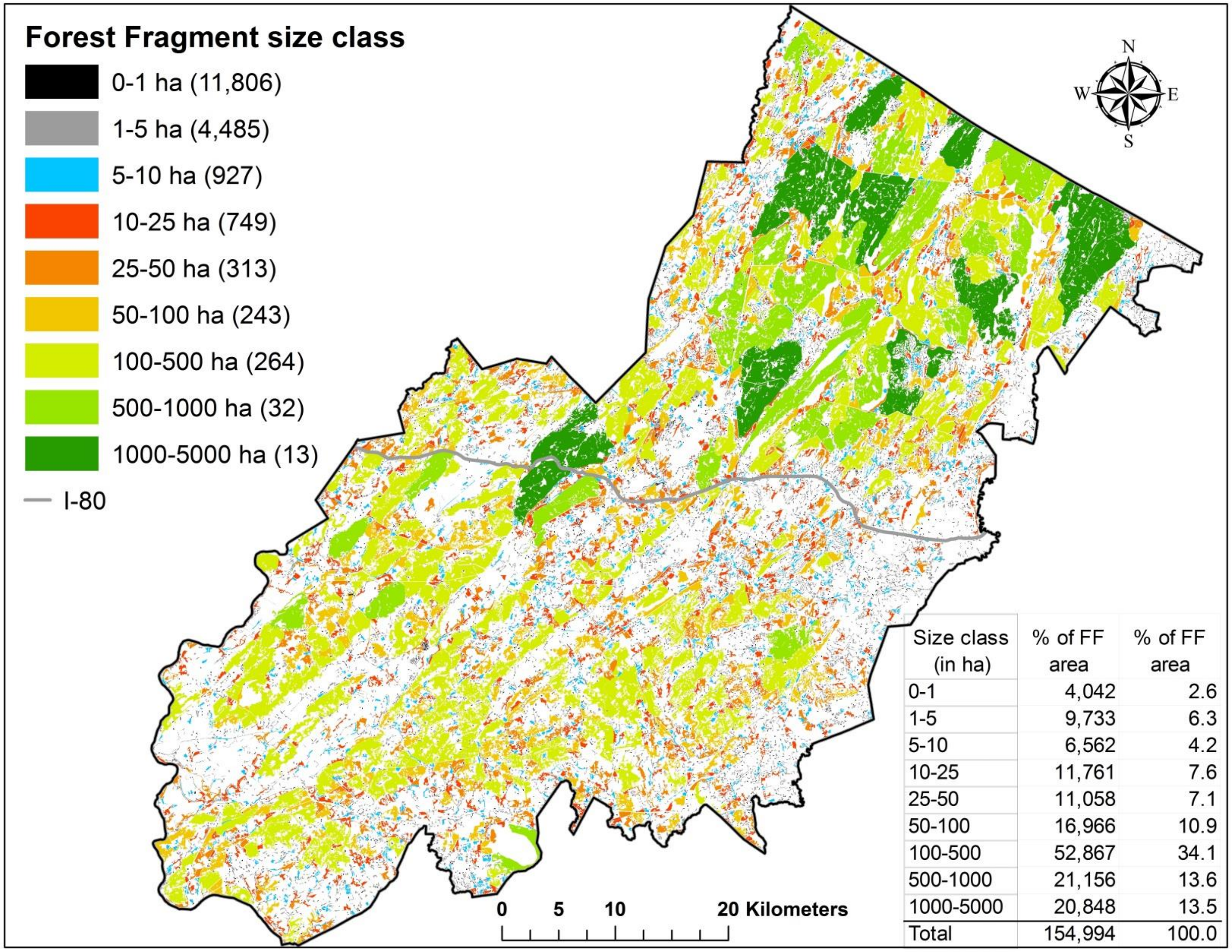


Figure 3.1a-c The figures illustrate the process for identifying the protected portions embedded within larger forests fragments and calculating the percent protection within each forest fragment. One or many individual tax parcels (shown by the grey lines in 3.1a) may compose a contiguous PF patch (the green region of Figure 3.1c). There may be one or many spatially distinct protected portions within a forest fragments (3.1c has three).



Size class (in ha)	% of FF area	% of FF area
0-1	4,042	2.6
1-5	9,733	6.3
5-10	6,562	4.2
10-25	11,761	7.6
25-50	11,058	7.1
50-100	16,966	10.9
100-500	52,867	34.1
500-1000	21,156	13.6
1000-5000	20,848	13.5
Total	154,994	100.0

Figure 3.2 The spatial distribution of forest fragments (FFs) of different sizes is mapped for the NJ Highlands. The count for each category is shown in parentheses. The inset table has the summed area for each size class.

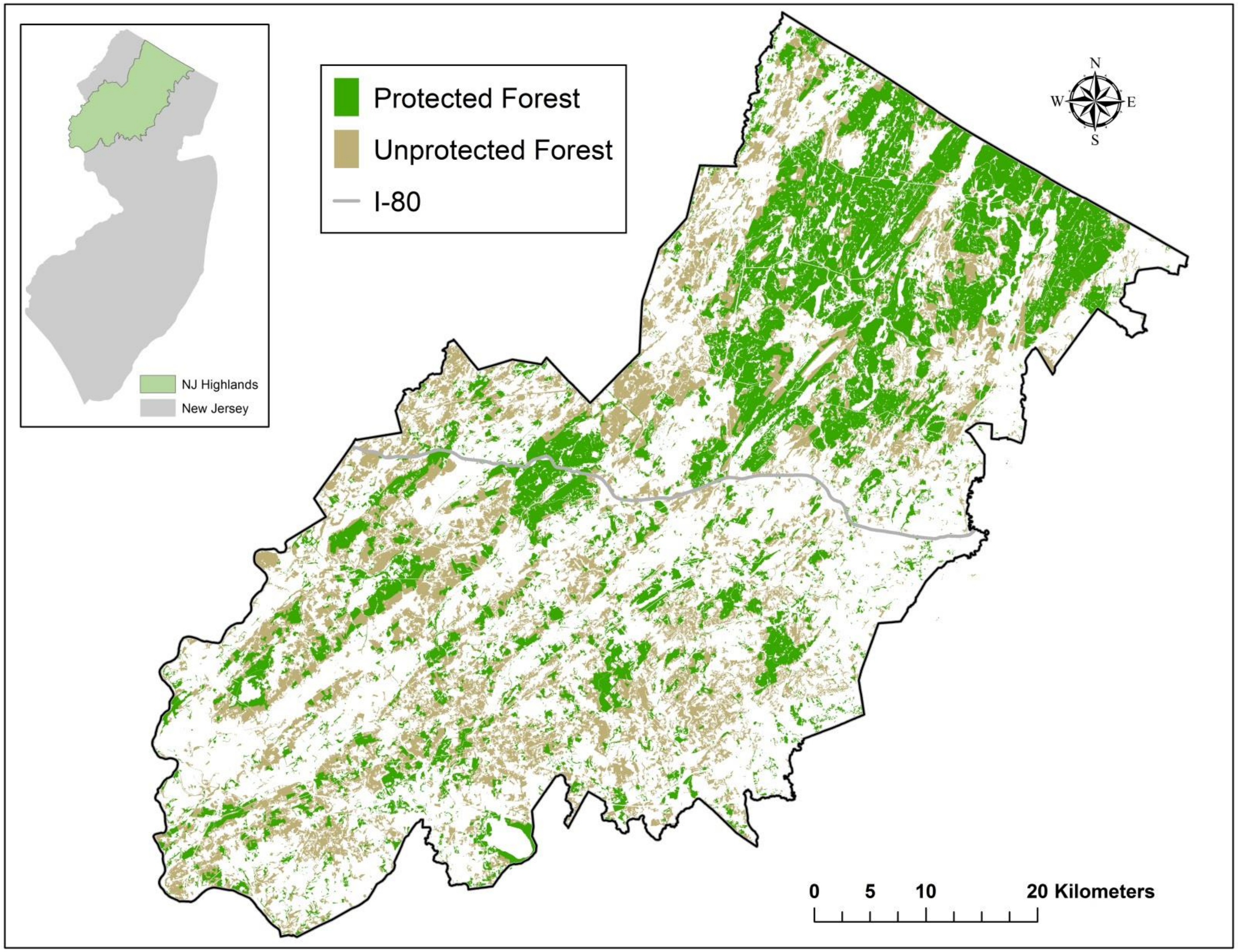


Figure 3.3 The location of unprotected forested lands (tan) throughout the Highlands are shown relative to the location of Protected Forests (green) which only partially occupy them.

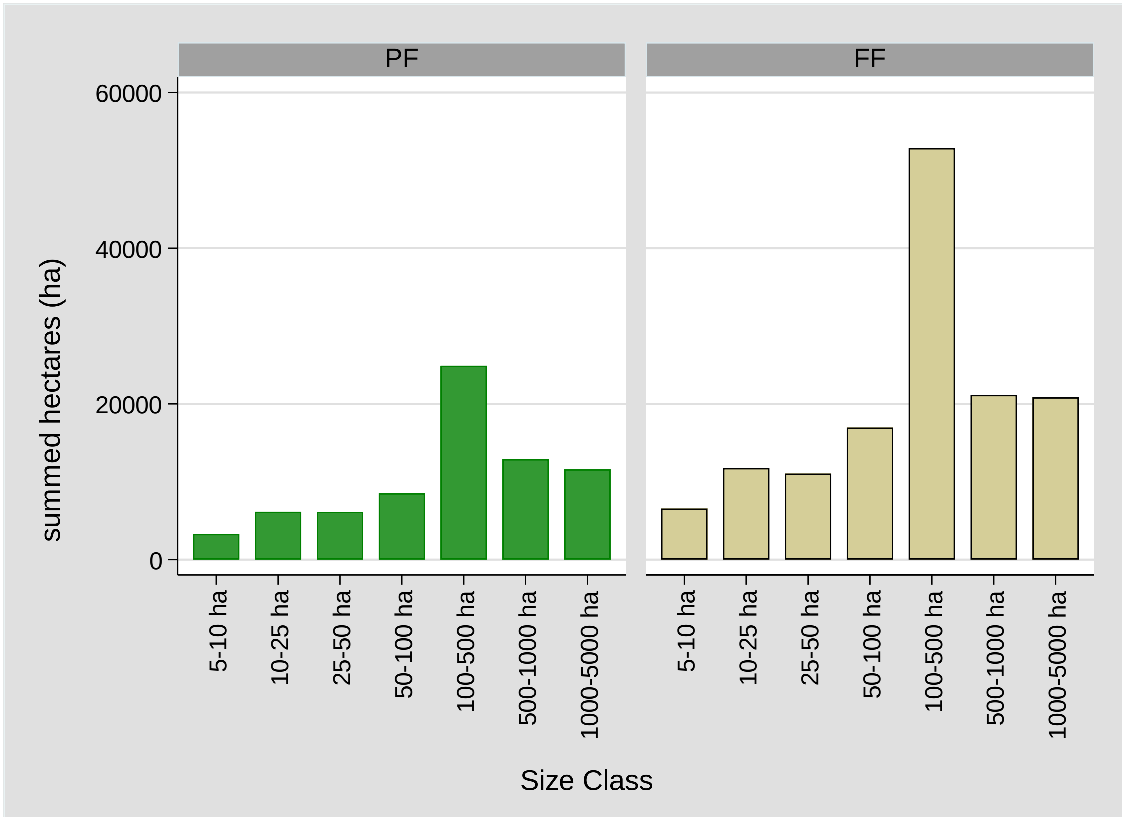
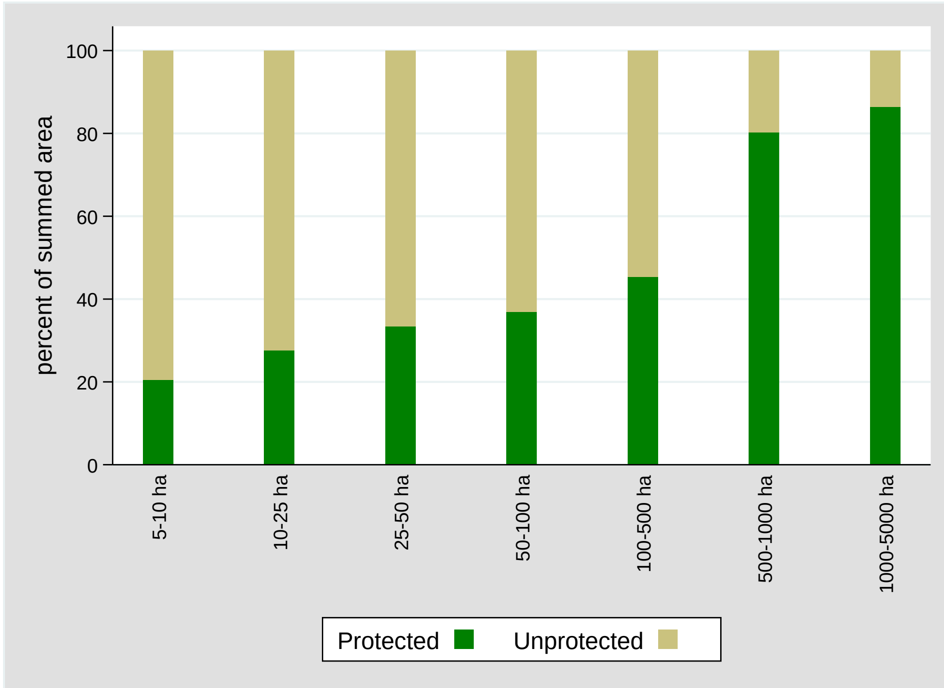


Figure 3.4 The distribution of the total area within Protected Forests size classes (a) compared to the distribution of the total area within Forest Fragment size classes (b).



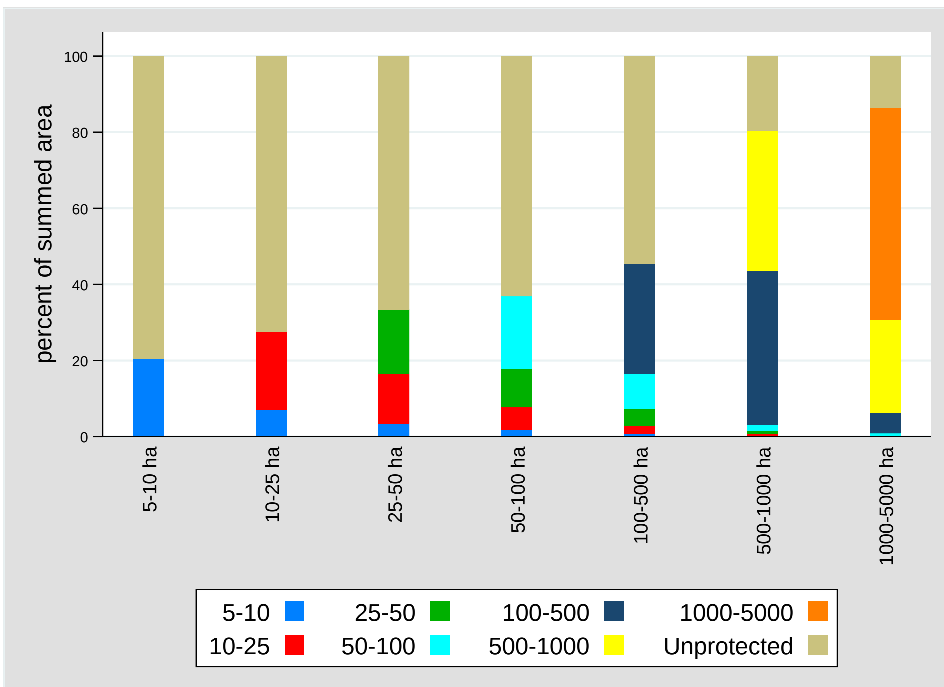
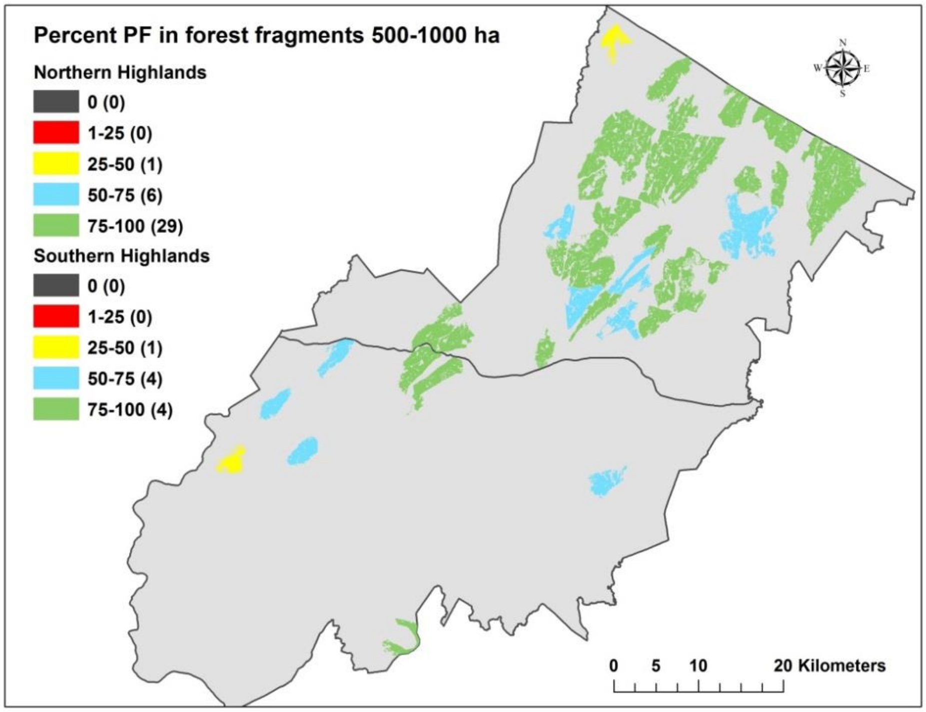


Figure 3.5a-b The size classes of FFs are shown along the x axis. The percent of land protected in each FF size class is shown on the y axis. The top graph sums Protected Forests of all size classes and represents percent protection in green. The bottom graph shows how much Protected Forests of different size classes (colored according to the legend in box) contribute to the total area protected within Forest Fragment classes.



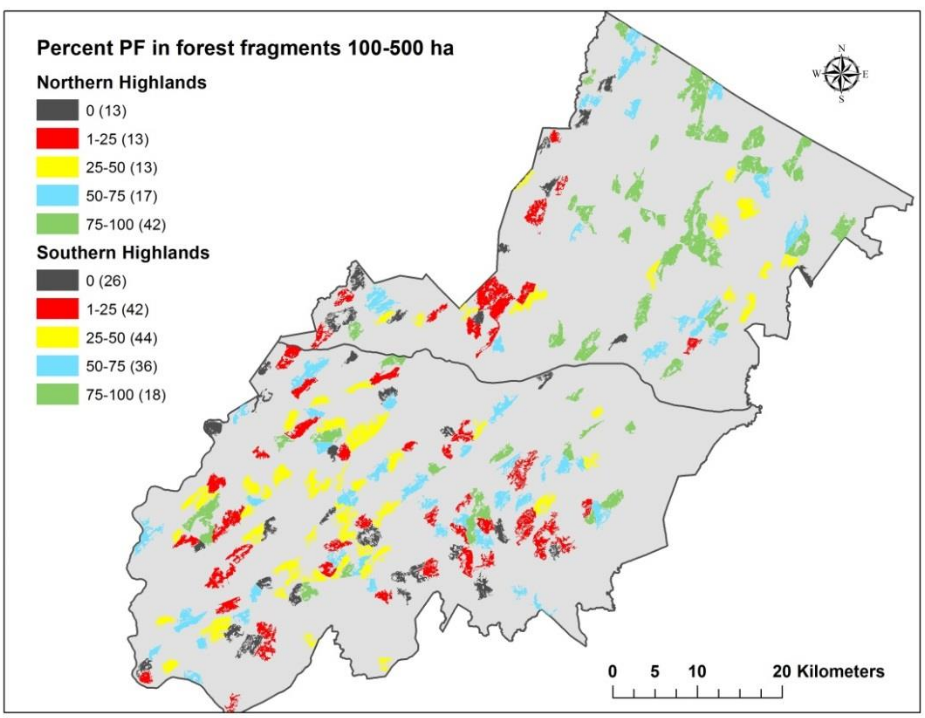


Figure 3.6a-b The spatial arrangement of all PF patches greater than 100 ha across the NJ Highlands, divided by the I-80 corridor. I-80 does not divide the Highlands landscape evenly (42% in the northern portion vs 57% in the southern portion), but roughly illustrates regions of good (south) and poor (north) farmland.

Chapter 4: Land cover change and its implications for enhancing edge contrast along Protected Areas boundaries

Introduction

Protected Areas (PAs) have become a cornerstone of biodiversity and ecosystem conservation efforts throughout the world. Increases in human populations and associated land conversions around Protected Areas (PAs) have been documented globally (Wittemyer et al. 2008) and been the focus of research in the U.S. (Radeloff et al. 2010; Wade & Theobald 2010; Hamilton et al. 2013). The relationship between human population density and biodiversity loss can be complex, but many studies document the negative impacts anthropic landscape modifications have on PAs. Wherever dense human settlement occurs, Protected Area size is often reduced (Luck 2007). Altering habitat near PAs enhances edge effects within protected habitat, disrupts the flow of biotic and abiotic resources between the PA and surrounding landscapes, reduces connectivity among PAs in a regional network (Hansen et al. 2005; Hansen & Defries 2007), These changes reduce survival of native species, increases exotic species occurrence in habitat remnants (Foxcroft et al., 2017) and alter patterns of regional diversity (Socolar et al., 2016).

Whereas deforestation is a primary cause of habitat fragmentation and isolation in many industrializing nations, residential and commercial development cause much of the habitat fragmentation in post-industrial nations (Radeloff et al. 2010). Throughout the U.S., rates of development are often higher around protected public lands than elsewhere in a region (Hansen et al. 2005; McDonald et al. 2007; Radeloff et al. 2010; Gimmi et al. 2011; Mockrin et al. 2013). This pattern holds even when proximity to urban areas is accounted for as an alternate amenity attracting development (Wade & Theobald 2010). Land development outside PAs also increases the likelihood of new development within

PAs (Leroux & Kerr 2013), presumably to add amenities for users. Although 'urbanization' occurs along gradients (McKinney 2002; Zipperer & Guntenspergen 2009) it can impact PAs anywhere along that gradient. Because land development and land conservation in the U.S. are often coupled (Beuschel & Rudel 2009; Rudel et al. 2011), the potential for ongoing and additional impacts from urbanization will likely increase rather than decrease. Thus, land cover transitions along PA boundaries have important implications for site management, if PAs are to function as cornerstones for conservation.

Interactions between Protected Areas and the landscapes around them

Conservation literature has highlighted the fallacy of applying Island Biogeographic Theory (IBT) to terrestrial systems management (Doak & Mills 1994; Laurance 2008) and proposed alternative conceptual approaches to guide biological conservation in fragmented habitats (McIntyre & Barrett 1992; Fischer & Lindenmayer 2006; Fahrig 2013). IBT was originally applied to terrestrial conservation problems because human land modifications appeared to create remnant habitat 'islands' in seas of human land use (Ewers & Didham 2006; Prugh et al. 2008; Saunders et al. 2009). However, experimental work has shown that IBT predications are often inconsistent with data from terrestrial environments (Laurance 2008; Prugh et al. 2008). Habitat fragment size and isolation, both of which are central to IBT, are poor predictors for species richness, abundance, and site occupancy in many land systems (Debinski & Holt 2000; Prugh et al. 2008; Resasco et al. 2017). Moreover, the predictive power of patch size and isolation shows variability across taxonomic classes (Debinski & Holt 2000).

Consequently, literature on conservation and management of PAs increasingly recognizes that the land cover and land use between habitat patches or conservation areas plays an important role in sustaining wildlife populations within those areas

(Lindenmayer & Franklin 2002; Prevedello & Vieira 2010). This interstitial space between protected habitats, often called the landscape "matrix", is typically associated with human land modifications. The matrix can be a heterogeneous mix of diverse land use changing rapidly over small spatial scales, or it can be more uniform across space. In addition to this spatial dynamic, land use has temporal dynamics. It change through time (e.g. a subdivision replacing an agricultural field) or remain relatively constant. In PA management, the matrix "matters" because it is not uniformly hostile to all species, but may support species movements, foraging, and breeding (Fischer & Lindenmayer 2007; Prugh et al. 2008; Saunders et al. 2009; Prevedello & Vieira 2010). A hospitable matrix can increase the likelihood of sustaining wildlife populations within PAs. Such support depends upon the type, amount, and arrangement of resources (such as food, water, or shelter) within the matrix.

Edge effects

In addition to supporting PA wildlife, the land use matrix can directly influence the quality of protected habitat by altering ecological processes along the PA boundary (Prugh et al. 2008; Saunders et al. 2009). At the legal boundary between protected and unprotected land, the land use and cover may show abrupt transitions creating a distinct edge or boundary between two cover types. For smaller PAs, the matrix becomes increasingly important because of edge effects. Edge effects are changes in biotic and abiotic patterns occurring at these transition zones (Forman 1995). As the size of habitat remnants decreases and shape becomes more irregular, remnants are increasingly dominated by edge habitat, making the ecological processes within these regions critical to both conservation and management decisions (Ries et al. 2004). Edge effects may extend from just meters up to a kilometer away from a land cover transition zone. Their extent and patterns depend, in part, upon the vegetative and topographic "contrast" at the boundary between two land cover types (Forman 1995; Lindenmayer & Franklin 2002; Fischer & Lindenmayer 2007). "Contrast" describes the amount of structural and compositional similarity across the transition zone and the spatial extent over which the transition occurs (Forman 1995; Ries et al. 2004). Unlike islands which necessarily have high-contrast edges where land and water meet, habitat contrast across the terrestrial landscape is highly variable. Croplands adjacent to woodland, for example, create a higher contrast edge than woodlands adjacent to grazed pasture (Fischer & Lindenmayer 2006).

Studies focusing upon the role of edge contrast are not as common as studies concerning the role of the regional matrix, but the two are inherently related and critical to habitat management decisions (Campbell et al. 2011). Research for different taxonomic groups has shown that permeability and use of the matrix by species often depends upon reducing the contrast along an edge or boundary. This dependency has implications for species conservation within developed landscapes. For example, Taylor et al. (2016) found that decreases in species richness for interior forest birds were not linked to the intensity of urbanization in the matrix surrounding a habitat remnant, but rather the reduction of tree cover within the matrix. Tree density, rather than grazing intensity, was also the primary driver for avian forest bird response and use of pastoral lands around remnant forest patches in Australia (Hanspach et al. 2011). Ikin et al. (2013) found that increasing amounts of native tree species (versus exotic species), and increasing habitat complexity made suburban landscapes more permeable to several groups of native forest birds.

Although considerably less well studied, other taxonomic groups show patterns similar to those of avian communities. In a study of small mammals, the "landscape element similarity" of the matrix (i.e. vegetative structure and composition) also played a role in shaping the permeability of the matrix (Brady et al. 2009). Like Taylor et al. the small mammal study argues that development intensity alone did not adequately explain habitat structure in the matrix. Residential lot size and landowner behavior both showed potential to modulate the effects of development intensity. Campbell et al. (2011) compared invertebrate communities along the edge of small (<100 ha) forest remnants embedded in both pastoral lands and pine plantations. The dominant driver of edge response (and hence turnover in invertebrate community composition) was structural contrast between the forest remnant and the landscape matrix. The low-contrast matrix (pine plantations) mitigated or nullified the edge effects observed in the high-contrast matrix (livestock pasture).

The studies cited above highlight nuanced and useful ways to understand human land use transitions near PAs. Collectively they suggest that, near PAs, landscape permeability should not only be evaluated through the lens of current land use, but also consider temporal dimensions of land use. Specifically, how does a given land cover transition increase or reduce the edge contrast and permeability relative to the land cover that it replaces, not just a pre-human or idealized natural land cover. The purpose of this study is to examine that question in greater detail.

Objectives: Why study PA boundary transitions

Regions where land cover transitions increase or decrease contrast along PA boundaries are sites of conservation interest. Such transitions produced a "two-sided" effect (Ewers & Didham 2006; Campbell et al. 2011; Schneider et al. 2014) essential to consider for effective conservation management. On the PA side, edge effect within protected habitat may be either reduced or exacerbated along sites of land cover transition. On the matrix side, the permeability of unprotected landscape may be increased or decreased for species of conservation interest. Changes in edge effects and matrix permeability are particularly relevant to conservation management in urbanizing regions where Protected Areas (PAs) tend to be smaller because of human settlement patterns (Luck 2007). Urbanizing landscapes like New Jersey, Florida, California, and Massachusetts have been popular sites for local open space funding via ballot measures (Lerner et al. 2007; Nelson et al. 2007; Szabo 2007). Through time, this funding mechanism has built PA networks with a wide range of habitat patch sizes and shapes adjacent to a diverse arrangement of land uses.

Throughout the northeastern U.S. a matrix of forest and farmland exists in many states of recent and historical transition. The region experienced extensive deforestation from the mid-1700s until the late 1800s, followed by a period of forest recovery as landowners abandoned farmland and other extractive industries (Ramankutty et al. 2006; Ellis et al. 2010; Lathrop 2011b). Since the 1970s, both farmland and forests have experienced a net loss due to increased demand for residential and commercial development (Riitters & Coulston 2005; Wickham et al. 2007; Drummond & Loveland 2010; Klepeis et al. 2013). However, the small scale of land transitions makes patterns of change in structural contrast at the boundaries difficult to discern through visualization alone. To understand the frequency and spatial distribution of structural contrast change, I analyzed land cover transitions from 1986-2012 along the boundaries of one specific type of protected habitat in this region: upland Protected Forests (PFs). I analyzed both individual sites and aggregate patterns by size class, focusing primarily on transitions to development because of its prevalence and implications for PA management (Theobald et al. 1997; Hansen et al. 2005; Radeloff et al. 2010; Gimmi et al. 2011; Hamilton et al. 2013).

My objective was to analyze how often development on PA boundaries replaces highly modified land (in my study area this is primarily agriculture) versus vegetative cover. In any land cover transition, land changes "from" one type "to" another. It is the transition itself, rather than just the final land cover which has implication for increasing or decreasing boundary permeability. Agricultural transitions into development, for example, have the potential to reduce the land cover contrast at PA boundaries if shade trees and landscape plants are planted, and vegetation along hedgerows and riparian corridors expands in the absence of intense tilling and livestock use. These combined changes may effectively 'soften' the edge (Forman 1995), making it more permeable to species. Conversely, urban development carved from former forest or vegetative cover often means a net loss of vegetative structure and complexity, a transition which should 'harden' boundaries making them less permeable to species.

Methods

Study site

The NJ Highlands is an urbanizing landscape. By 2000, 24% of the NJ Highlands was classified as "developed land", a classification including residential, commercial, and industrial land use at all densities. However, the southern NJ Highlands still support significant amounts of 'prime' farmland, often a mixture of cropped land and dairy pasture (Phelps & Hoppe 2002). From 1984-2000, regional losses of forest and farmland occurred at rates comparable to one another, with forest/wetlands losing 33,877 acres (13,710 ha) and farmland/grassland losing 32,590 acres (13,189 ha)(Phelps & Hoppe 2002). This period of loss was coupled with a 65,570 acre (26,535 ha) increase in development all along the urban gradient. Because of this, the NJ Highlands are a good location to study how urbanization replaces both forest and farmland along PA boundaries and changes landscape permeability.

GIS processing

I used the PATX dataset (Chapter 2) to identify all Protected Forest patches greater than five hectares (n = 1287). Upland Protected Forest (PF) "patches" were previously spatially defined using just the upland forested habitat within contiguous protected tax parcels (Chapter 2). The implications of excluding wetland forests from the spatial extent of PF patches are addressed in the Discussion section of this chapter. I buffered each PF patch at 250 meter (m). Land transitions at this scale have direct impacts upon adjacent forest edge vegetation and processes (Alverson et al. 1988; Roland 1993; Weathers et al. 2001; LaPaix et al. 2012). Any transitions within this 250 m zone would potentially alter the edge effects and permeability at PF boundaries. I assigned each buffer a unique ID number to link their attributes back to their PF patch for mapping and analysis.

I intersected the buffer polygons with the 1986 and 2012 land use/land cover data (lulc) (NJDEP 1986, 2012) to obtain the unique land cover mosaic within each individual PF buffer. (To distinguish this analytic unit from the broader landscape matrix occurring between PFs, I refer to this as the "buffer mosaic" throughout the text.) From the land cover data, I identified buffer mosaics in each time step and analyzed their transitions at two levels. 1) I mapped and quantified the boundary transitions for individual PFs. 2) I aggregated boundary transition data across the seven largest PF size classes. NJ land cover datasets are vector-based with a minimum mapping unit of one acre (0.405 ha). Thus they showed fairly detailed data on land cover even though buffers were only 250 m wide. Many transitions occur at the parcel level as land owners change management practices so this level of detail is important in an urbanizing landscape.

I classified buffer mosaics using the modified Anderson Level 1 classifications within the lulc dataset (Chapter 2). Level 1 consists of six possible land cover types: Agriculture, Barren, Urban, Forest, Water, Wetlands (NJDEP, 2002). Buffer mosaics were defined using three general land cover types (Agriculture, Developed, and Natural) at the boundaries of PF patches. I collapsed the Level 1 Forest, Water, and Wetlands classes into the "Natural" cover class. All Urban patches I reassigned as "Developed". Agriculture did not change. The "Barren" category indicates land cover in transition with an absence of any cover type. I omitted Barren lands from the analysis because 1) they were less than 1% of a given buffer, and 2) they could not properly be assigned to a class. All spatial computations were done in ArcGIS software version 10.2. (ESRI, 2014)

Buffer mosaic classification

To classify each buffer mosaic, I adapted a land cover classification scheme published by Riitters et al. (2009). This scheme represents three dimensions of land cover (Agriculture, Developed, and Natural) in a two-dimensional space using a tripolar or ternary plot (Figure 4.1a). The plot provides a visualization of the relative percent each of the three land cover types contributes to the overall mosaic. That three-part mosaic is also represented through the naming scheme for each class. The letters "A" or "a", "N" or "n", and "D" or "d" represents Agricultural, Natural, and Developed land cover, respectively. Capital letters indicate that a polygon (in this analysis, the 250 m buffer) has at least 60% and up to 100% of any given land cover. Small letters indicate that a polygon has at least 10%, but less than 60% of any given cover. The absence of a letter indicates less than 10% of that cover in the polygon. Percentages of one cover type are always given relative to the others. Thus, if the percent of one land cover type increases, another necessarily decreases.

Whereas Riitters et. al. divided their tripolar plot into 19 different land mosaic classifications (Figure 4.1a), I reduced this to seven (Figure 4.1b). This simplified reporting transitions in the buffer mosaic over two time steps. To reduce the number of classes to just seven, I consolidated any class with 60-100% of a given land cover type into simply “A”, “D” or “N”. This merged five of the Riitters et al. land cover classes into a single one at each angle in the tripolar plot (Figure 4.1b). The “dn”, “an”, “ad”, and “adn” categories of Riitters et al. did not change in my adapted classification. These latter categories represented buffers composed of two or more land cover types, each with a given land cover of at least 10%, but less than 60%.

Individual site analysis

I classified the buffer mosaics in 1986 and in 2002 for each of 1287 PF patches, and passed that attribute back to my PF layer. I mapped all PF patches, coding each according to the individual patch’s buffer mosaic. From this, I identified all PFs with a classification change from 1986-2012 (e.g. the buffer was “D” in 1986 and “dn” in 2012) and created a binary map of the results (no change = 0, change = 1). All individual transitions from one buffer class to another were enumerated and reported by PF size class. This analysis identified how many individual PF patches might experience future impacts and management challenges from land change on their borders. I defined the total area of changed buffers as the summed area of all buffers in each size class which changed over the study period. I report on both the count of transitions that occurred in each size class, and the total area of changed buffers within each size class.

Aggregated size class analysis

I aggregated the buffer mosaic data by year and size class for each of seven PF size classes (i.e. 5-10 ha, 10-25 ha, 25-50 ha, 50-100 ha, 100-500 ha, 500-1000 ha, 1000-5000 ha). These were plotted onto a tripolar graph for each study year using R package ggtern (Hamilton 2018). Plot points for each size class were determined by summing the total buffer area for a given size class, and calculating the relative amount of Agriculture, Developed, and Natural land within that summed area. This permitted me to assess 1) if different PF size classes were characterized by different buffers, and 2) the rate and direction of buffer mosaic change for each size class between the two study years. If buffers within a size class overlapped one another, the buffer area would have been counted twice in the process of calculating relative percentages. This was deliberate as I wanted to account for all the land cover surrounding each individual PF.

Results

PF buffer mosaic classifications

The buffer mosaics for PFs fell within six of the seven possible land cover classes in both 1986 and 2012. None fell into land cover type "ad" (dominated by agriculture and developed land). The most common buffer mosaic was type "N" which bounded more than 700 individual PF patches (representing ~60,000 ha of PF) in both years (Figures 4.2 and 4.3). The second most common buffer mosaic was "adn" which bordered over 200 PF patches (~6000 ha of PF) in both years. The remaining buffer types bordered 100 or fewer PF patches in both years, with the single exception of the "dn" class. This class bounded 94 PFs in 1986 (~2300 ha), but 170 PFs in 2012 (~6000 ha). Between the two time steps, the count of three buffer types increased ("D", "dn", and "adn") and counts of three others decreased ("N", "A", and "an"). Interestingly, although the count of "adn" buffers increased from 226 to 252, the total area of PF forest bounded by these buffers decreased from 6861 ha to 6122 ha. This suggests that "adn" increased around smaller PFs and decreased around some larger ones. This was the only buffer class in which a decrease or increase in count did not signal a corresponding increase or decrease in the total PF area bounded.

The location of PFs with changed and unchanged buffers is mapped in Figure 4.4. Out of 1287 individual PFs, 283 had a buffer classification shift between 1986 and 2012. Buffer mosaic changes were more common in the southern portion of the study site and PF patches with changed and unchanged buffers often occurred in close proximity to one another. Their close proximity indicates the spatial heterogeneity of land change throughout the region, and that land change often occurred at small spatial scales.

Analysis of change by PF patch size: count and summed hectares

Table 4.1 shows the summed area of changed and unchanged buffers for each size class, along with the count of PF patches with changed and unchanged buffers. The majority of individual PF patches within each size class did not experience a buffer mosaic change. When change occurred, it was disproportionately represented in the smaller size classes. In the 5-10 ha size class, buffer mosaics changed around 135 patches (29% of patches). For the 10-25 ha size class, buffers changed around 81 patches (20%). In the 25-50 ha class, buffers changed around 35 patches (20%). In the 50-100 ha class, change occurred around 21 patches (17%). In the 100-500 ha size class, nine patches (8%) had buffer change. In the 500-1000 ha size class, change affected two patches (8%). All seven patches in the largest size class had stable buffers from 1986-2012.

Total changed buffer area ranged from 972 ha in the smallest size class, up to 1872 ha in the 500-1000 ha size class. The small range of buffer changes across size classes was unexpected given that the total area (the sum of changed + unchanged buffer area) was as small as 3300 ha in the 5-10 ha class, but as large as 24,907 ha in the 100-500 ha class. For the smaller size classes, up to 30% of the total buffer area changed. Thus, the PFs most vulnerable to edge effects, due to their small size, were also those most likely to have experienced a change in their buffer during the study period. However for the 100-500 ha class, which contains more PF land than any other class, changed buffers accounted for only about 6% of the total buffer area.

Buffer mosaics for individual PF patches: "from - to" land cover transitions

A count of individual PFs with each type of buffer mosaic in 1986 and 2012 are given in Appendix B as a series of transition matrices. The mosaic class from which a buffer transitioned in 1986 is read from the first column. The class into which it transitioned in 2012 is read from the second row. Values along the diagonal (in grey text) show the count of PF buffers with no transition during the study period. The most common individual transition in four size classes (5-10, 25-50, 50-100, 100-500), was a shift from "N" to "dn" (n=27, n= 9, n=7, n=4, respectively). For the 10-25 ha class, the most frequent transition was from "an" to "adn" (n=18). In the 1000-5000 ha size class, the only changes were "N" to "dn" (n=1) and "and" to "N" (n=1), so neither transition was more common. Overall, the transition of other cover into Developed was the most salient trend at PF boundaries across all size classes.

Summarizing other common transitions was difficult because they varied across size classes, but some general patterns can be discussed. Wherever buffer mosaics transitioned into "N" or added "n" to their classification label, the amount of natural land increased. Such transitions have the potential to decrease or 'soften' boundary contrast along that PF patch, thereby increasing permeability. These transitions are shaded green in the tables of Appendix B to indicate their impact on structural contrast. Wherever buffer mosaics transitioned into "D" or "A" from a prior category with "N" or "n", structural contrasts along PF boundaries would arguably have been increased or 'hardened'. These transitions are shaded grey to indicate their impact on structural contrast. The latter transitions (from “N” or “n” into “D” or “A”) occurred more often than the former in almost all size classes. However, the addition of natural land at PF boundaries was well represented throughout the mosaic transitions in size classes under 500 ha. Particularly for PFs under 50 ha, the tables in Appendix B show how dynamic the buffer mosaics were even over just a few decades. Other transitions (“an” to “adn”, and “adn” to “dn”) are also well represented in the two lowest size classes. I omit them from further discussion because their impact on boundary contrast is difficult to interpret unless assessed on a case-by-case basis to determine if “d” increases at the expense of “a” or “n” in the buffer..

Differences in buffer mosaics by size classes (within-year comparison)

In 1986 and 2012, the aggregate buffer mosaics for all size classes, except one, fell within thresholds for the “N” mosaic class (Figure 4.5). The 5-10 ha size class had just below 60% natural cover and therefore fell into the “adn” class. Buffer composition for all size classes ranged between 55-90% Natural, 10-30% Urban, and 0-25% Agriculture. Despite falling within the same “N” classification, most size classes had distinct buffer compositions and did not overlap one another. An exception to this was the 10-25 ha and 20-50 ha size classes, which were quite similar in both study periods. The two largest size classes also were also quite similar in composition in 2012, but became slightly more distinct in 1986.

Comparing buffer mosaics for each size classes within a given study year (that is, patterns within just the black circles or just the triangles of Figure 4.5) showed that aggregate Natural cover percentages increased as size class increased for both years. Agricultural percentages showed the opposite pattern; percentages increased as size class decreased. Urban percentages were similar for size classes less than 500 ha in both years

(~20% in 1986 and ~25% in 2012). Urban percentages were ~ 15% for PFs over 500 ha in both years.

Shifts in buffer mosaics by size classes (between-year comparison)

From 1986 to 2012, most size classes had a slight decrease along the Natural axis, coupled with an increase on the Urban axis (both less than 5%, Figure 4.5). These patterns were more pronounced in the size classes from 50- 500 ha. Agricultural change showed mixed results. The smallest size classes had up to 5% Agricultural decrease, but other classes had so little agriculture in 1986 that they showed little change. These collective results suggest that, in aggregate, buffer mosaics across most size classes accumulated developed land at similar rates, but the source of that increase was not the same. In the three smallest size classes, Developed values appeared to increase at the expense of Agriculture. Size classes above 50 ha increased Developed cover at the expense of Natural cover. The ~5% increase in Developed for most size classes does not mean that development was evenly distributed across individuals PFs in a size class. A small number of individual PFs may have gained a high percent of development in their buffers while others gain none.

Discussion

Boundary contrast increased overall, but many individual PF boundaries had contrast decreases

The goal of this analysis was to consider how shifts between past and present land use might enhance or decrease existing structural contrast at the boundaries of Protected Areas. Not all urban conversions are functionally equivalent because they replace different types of land cover. For example, distinguishing Agriculture-to-Development transitions versus Natural-to-Development transitions is of conservation interest. The former transition has the potential to soften vegetative contrast at the boundary through the introduction of landscape plants, shade trees, and encroachment of vegetation from formerly maintained hedgerows and riparian corridors. The latter transition would harden boundaries by increasing isolation from similar habitat.

Results from the aggregated data analysis showed that most size classes decreased in their percentages of natural land cover and increased in percentages of developed land cover. Because development often replaced natural land cover, PFs of this region overall experienced enhanced structural contrast or a "hardening" of borders between protected and unprotected habitat. This aggregate pattern was also supported in the data analysis on individual PF buffer transitions (Appendix B) where loss of natural land cover to development was a dominant transition in many size classes.

Protected forest patches are often contiguous with unprotected forest (Chapter 3). Where this occurs, the legal boundaries of PFs may shape the physical habitat boundaries as well. This can occur because legal boundaries effectively separate spaces that can be developed from those which cannot. As development occurs in unprotected forest, the legal, and formerly invisible boundary, of a PF becomes a visible boundary marked by a change in land cover. My results suggest that over the time period studies, this occurred around many of the Highlands PFs. This pattern was more consistent in the larger PF size classes. Such patterns of land cover change enhance edge effects within protected forests, and reduce permeability at the PF boundary. However, when the change occurs along larger PFs, a lower percent of the total PF habitat is impacted. Although reducing boundary contrast would be a desired goal for any size of PF in the Highlands, larger size classes should have more capacity than smaller one to buffer the impacts of development on PA communities.

Analysis of the individual buffer mosaic data also showed that agricultural transitions played a more complex role at PF boundaries than I had anticipated. In fact, there were no direct transitions from agriculture to development (or “A” to “ad”). Most transitions from agricultural cover involved shifts into both natural and developed cover. Although prior studies have shown that forest gains from agricultural abandonment are diminishing at regional scales here (Riitters et al. 2002; Drummond & Loveland 2010), my results show that such changes still occurred at small scales, particularly around the smallest PF size classes. These small gains have positive implications because these PFs are widespread and numerous. Where natural cover was added, edge effects could be mitigated and permeability at the boundary increased. Although PFs under 50 ha contribute a small percentage to the total area of Protected Forest regionally (Chapter 3), they can enhance connectivity between larger PFs (Shafer 1997). As such, it is important to focus on the contributions of these sites. They are not uniformly becoming more isolated, but show some evidence of favorable land cover change within their buffers.

My data only cover roughly 25 years, but show that urbanization is a temporally dynamic process around PFs, and that not all urban transitions might impact PFs in the same way because they replace different land cover types. In this case study, loss of agriculture to development frequently accompanied increases in natural habitat and therefore was not functionally equivalent to the potential impacts of a Natural-to-Developed transition around PFs. For studies focused on how urban development increases impact protected habitat, incorporating the land use prior to development into research may help explain confusing or contradictory interactions between protected habitat and developed landscapes.

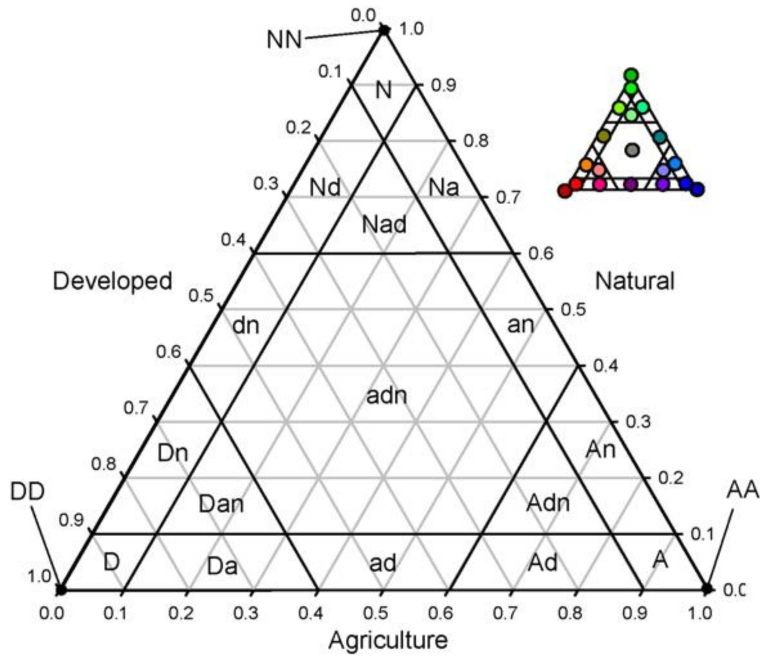
The focus of this analysis has been upland forests. As noted in Chapter 2, many wetland forests are contiguous with upland forests and could be included to expand the definition and size of the PFs analyzed. As a consequence of appending wet forest onto upland PFs, many would increase in area and therefore become members in a larger size class. The addition of wetland forests would therefore decrease the number of PFs in class sizes under ten hectares and increases the numbers in classes above ten ha (Appendix A).

Appending wetland PFs onto upland PFs could potentially change the results presented in this chapter because PFs would shift groups. The Agricultural-to-Developed transitions most apparent in PFs less than 50 ha might merge with the larger size classes and make these patterns less distinct. However, it is also possible that appending wetland to upland forests would simply enhance the spatial patterns of change already presented. By expanding the boundaries of a PF, more PF boundaries might then "bump up" against development. This expansion could then reduce the percent of Natural land I calculated around PFs, and increase the percent of Developed lands calculated. Given this, the types of land cover transitions would not necessarily change. Natural to Developed transitions would still figure prominently, but these existing patterns might just become stronger if wetland forests were included in the spatial extent of PFs.

The difficulty of classifying landscape mosaics near PAs

In addition to the findings above, this study offers a useful approach to classifying somewhat 'messy' landscape mosaics in urbanizing regions where land use changes over small spatial and temporal scales. Mapping sites by buffer mosaics offers a visualization through which land managers at individual sites can identify other PAs with similar configurations (i.e. size and buffer matrix) and assess how unique their management challenges are, relative to others in the region. Having a regional classification system can facilitate information exchange on management practices, collaboration on grants, and a more coordinated approach to management. NJ and other urbanizing sites offer a good laboratory for evaluating many management strategies because of the variability in PA size and landscape context. As urbanizing landscapes become more common, it is important to recognize them as information-rich sites for conservation studies.

As conservation literature focuses on understanding how the size and shape of habitat remnants interact with human land use in the matrix surrounding them, the most informative studies often report on idealized sites. This focus is critical to advance theory, but often fails to address how dynamic and variable land cover transitions are around most real-world PAs. The gap between characteristics of a site from a published study and those common on the landscape can be large, leaving many management problem unsolved even as theory advances. Although this study proposes a tool for classify landscape mosaics, it simultaneously illustrates the difficulty of doing this. Completing a detailed classification of transitions around PAs does not make for easily interpreted results. However it does highlight the complexity of PA management challenges that academic studies still need to address.



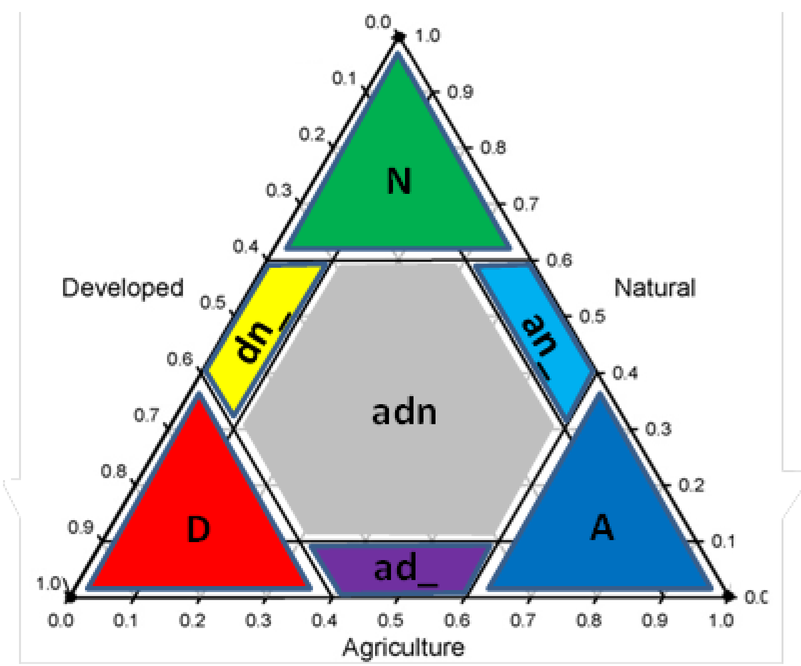


Figure 4.1a-b (top) Riitters et al.'s original tri-polar chart illustrating the landscape mosaic classes derived from the proportions of Developed, Agricultural, and Natural land-cover. The inset chart shows the colors used to render maps of landscape mosaics based on the classifications (a). *From Riitters, K. H., Wickham, J. D., & Wade, T. G. (2009). Ecological Indicators 9: 107–117*
(bottom) The modified classification scheme I used to categorize the land composition of individual PF buffers. This collapses several categories into more general one to simplify reporting of land transitions in PF buffers (b).

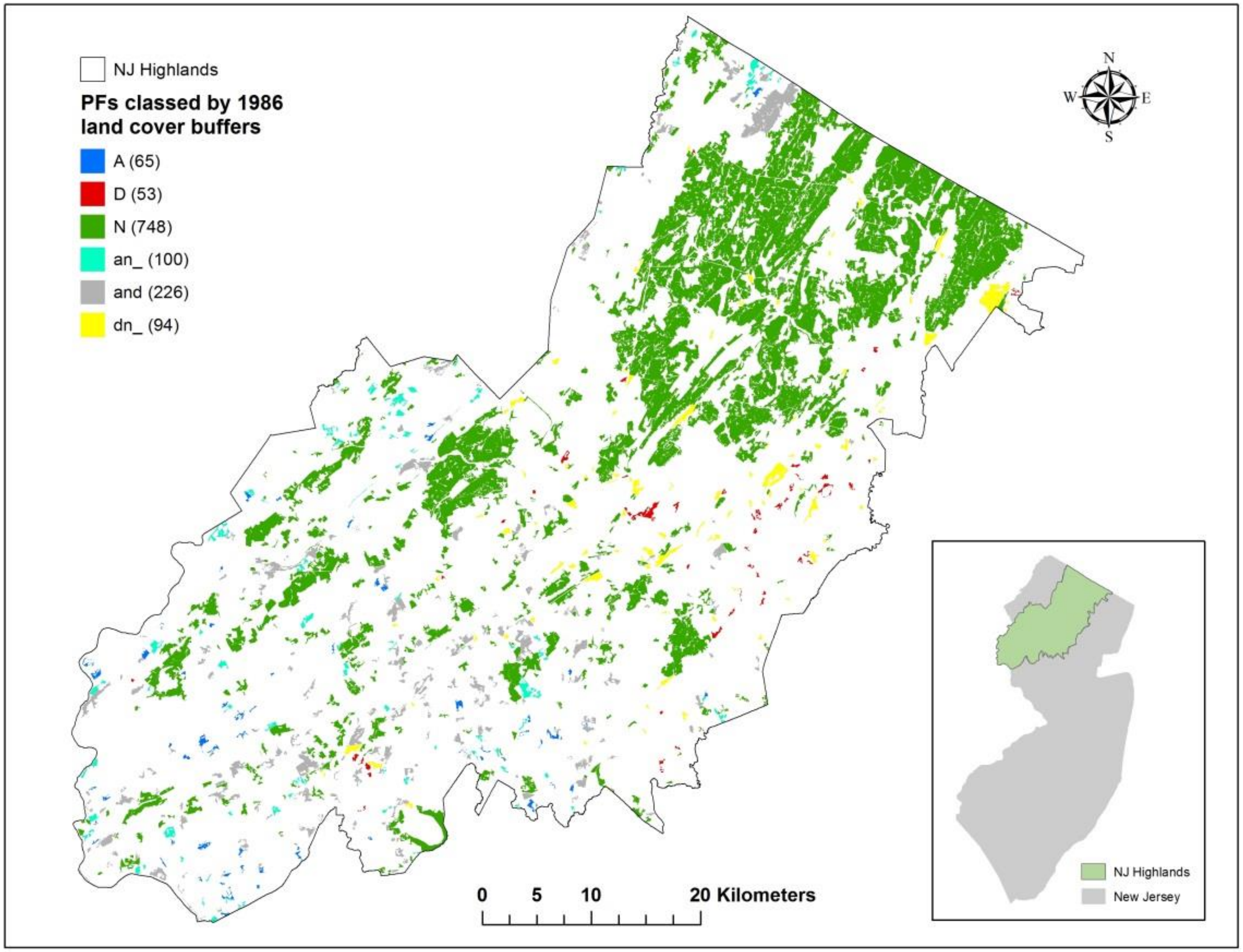


Figure 4.2 Protected Forests classified according to the 250 m land cover buffers surrounding them in 1986. “N” was the most common buffer mosaic and was associated with many of the largest PF parcels in the northern Highlands.

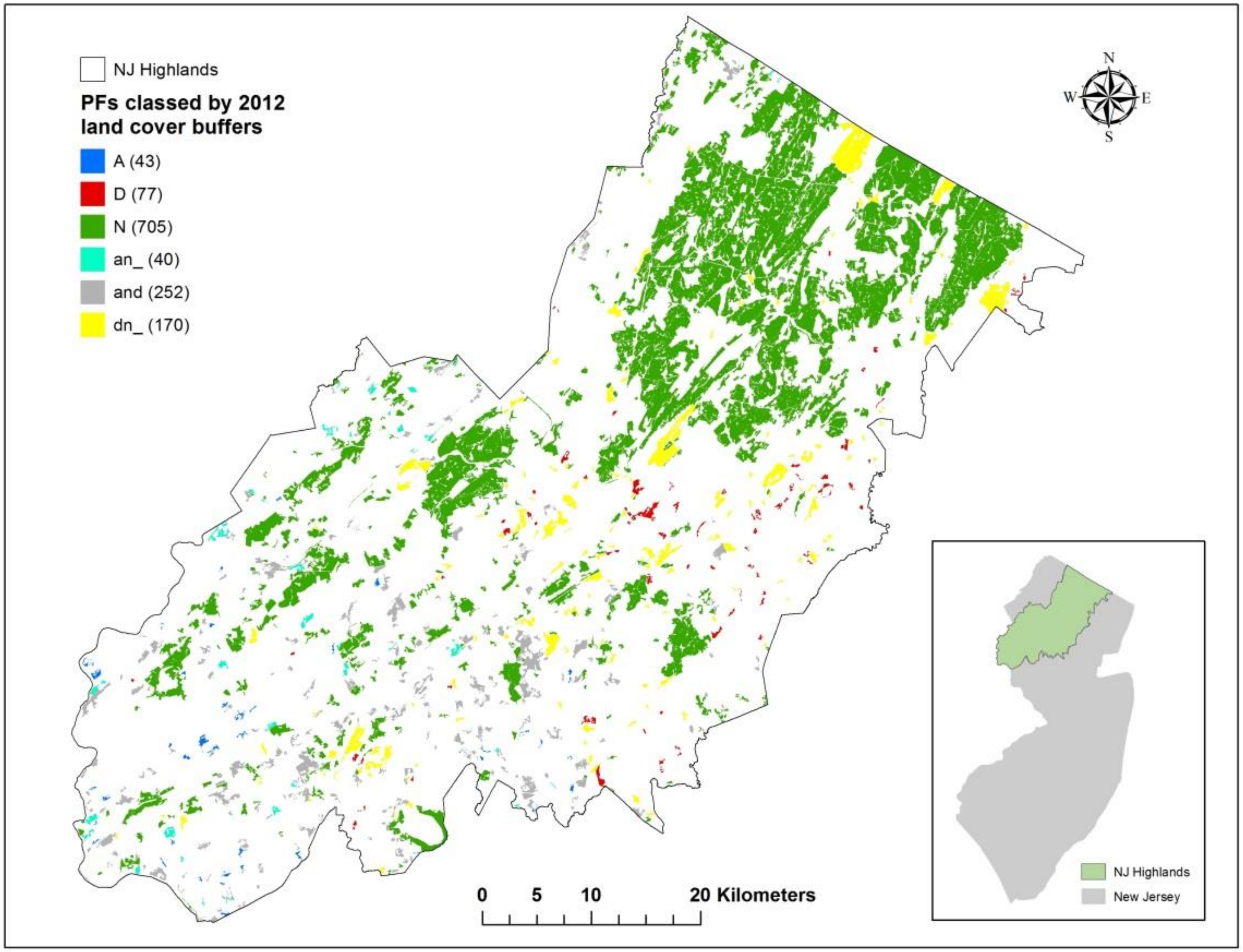


Figure 4.3 Protected Forests classified according to the 250 m land cover buffers surrounding them in 2012 (n=1287).

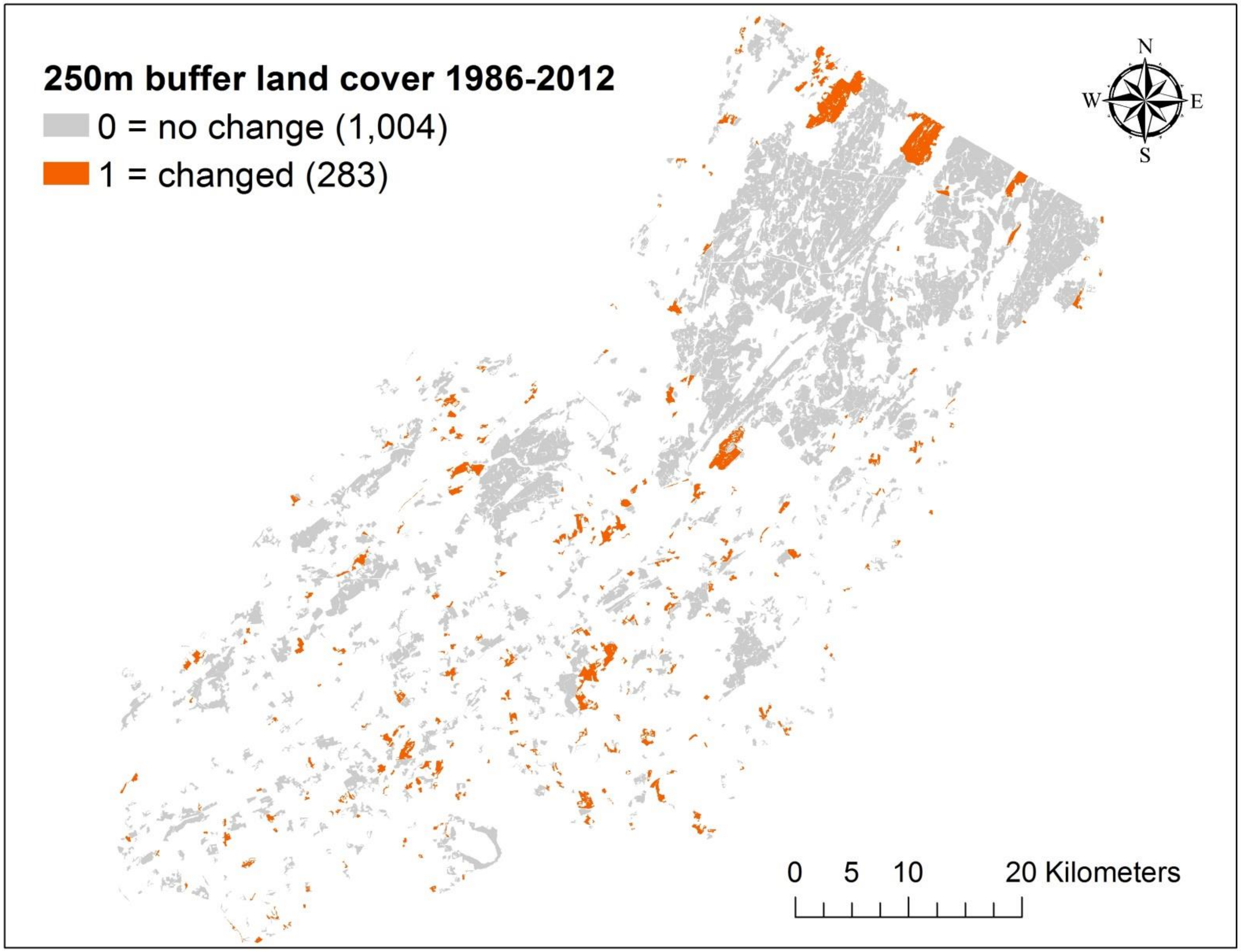


Figure 4.4 Protected Forests patches for which the buffer mosaics changed between 1986 and 2012.

Table 4.1 Summary data on the buffer mosaics which changed land cover class between 1986 and 2012 are given below, subdivided by size of PF patch The total amount of land (in hectares) is shown at the top of each cell. The count of PF patches are in grey at the bottom. All area amounts are rounded to the nearest hectare (n=1287 PF patches).

Size Class	No Change (ha) (count)	Change (ha) (count)	Total (ha) in buffer
5-10	2328	972	3300
	(325)	(135)	(460)
10-25	4881	1271	6152
	(316)	(81)	(397)
25-50	4899	1244	6143
	(144)	(35)	(179)
50-100	7207	1314	8521
	(100)	(21)	(121)
100-500	23,249	1658	24,907
	(97)	(9)	(106)
500-1000	11,031	1872	12,903
	(15)	(2)	(17)
1000-5000	11,600	0	11,600
	(7)	(0)	(7)
Total ha	65,195	8331	73,526

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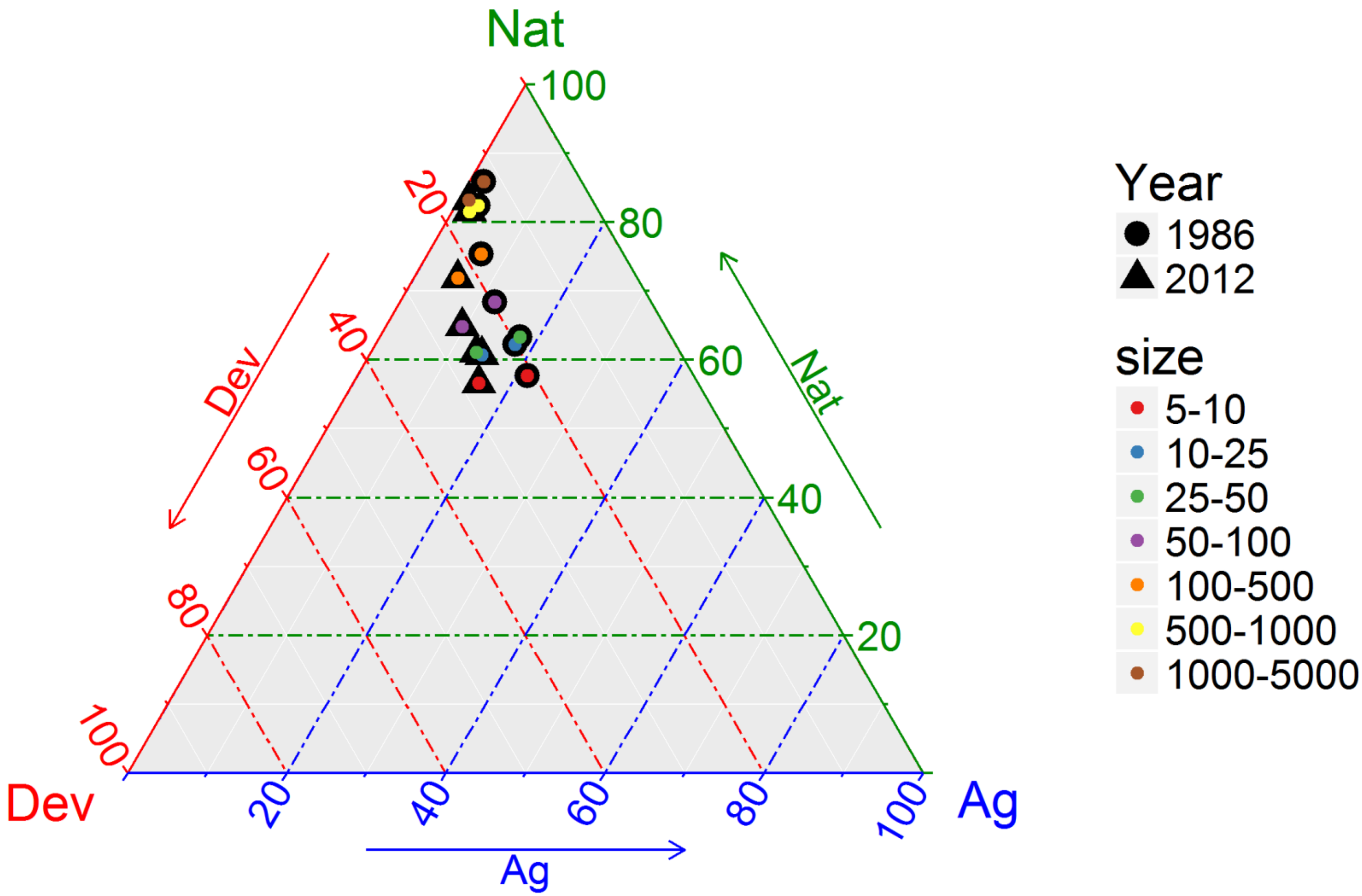


Figure 4.5 Tripolar chart showing the relative amount of three land cover types surrounding PFs parcels of different size classes.

Chapter 5: The prevalence of proactive versus reactive land conservation strategies in the New Jersey Highlands

Introduction

The amount of legally protected land across the globe has risen dramatically in recent decades (Chape et al. 2003; Zimmerer et al. 2004; Dudley 2008). Within the U.S., this corresponds to an increase in small land acquisition agencies and their growing role in land conservation (Merenlender et al. 2004; Lerner et al. 2007; Davies et al. 2010; Armsworth et al. 2012). When land parcels are given protected status, it often gives the impression that the site has high natural resource value. Otherwise, why protect it? However, the practice of protecting land is motivated by goals as diverse as the agencies and actors involved. Some agencies focus on land conservation within specific administrative boundaries (Nelson et al. 2007) whereas other operate nationally with biodiversity as a primary focus (Armsworth et al. 2012). Consequently, although significant amounts of land now have protected status across the U.S., not all of it may be high quality or high priority land for natural resource conservation purposes.

Legally protected lands are often generically referred to as "open space" in the U.S. and may be established independent of any formal biodiversity conservation plan. There is a substantial and evolving body of literature on Systematic Conservation Planning (Margules 2000; Sarkar et al. 2006), a process which yields formal conservation plans. This approach typically incorporates land use and species presence data into computer algorithms to select land parcels for conservation under dynamic economic and ecological constraints. Systematic Conservation Planning (SCP) seeks specific solutions to either minimize species loss or maximize species protection in dynamic landscapes.

The gap between land acquisitions priorities proposed within SCPs and the land that is actually acquired has been highlighted by several authors (Prendergast et al. 1999; Knight et al. 2008; Carter et al. 2014). The process for protecting land can be complex and may not address or incorporate high priority sites because it interfaces with social and economic landscapes, not merely environmental ones. Institutional capacity, opportunity, and land markets all play important roles in whether a land acquisition agency actively engages with any internally or externally developed SCP (Fishburn et al. 2013; Carter et al. 2014, 2015). Acquisitions in the United States rely heavily on the confluence of a willing seller, a willing buyer, and the availability of funds (Fairfax et al. 2005). Acquisition decisions are often time-sensitive because windows of opportunity, such as funding or a landowner's willingness to sell, are limited (McDonald-Madden et al. 2008). Furthermore, these decisions occur in dynamic land markets in which 1) individual land use decision can feed back into land costs (Armsworth et al. 2006; Chamblee et al. 2011; Dissanayake & Önal 2011; Butsic et al. 2013) and 2) the goals of individual institutions are constantly evolving (Fishburn et al. 2013).

The selection of specific parcels for protection is increasingly the domain of many small non-governmental organization (NGO) land trusts, as well as local and regional governments. Local government's growing role in land acquisition stems from the legal authority established via ballot measures (Kotchen & Powers 2006; Lerner et al. 2007; Nelson et al. 2007; Szabo 2007). New Jersey municipalities and counties have shown especially strong support for such ballot measures (Myers 1999; Solecki et al. 2004; Heintzelman et al. 2013). For land protection entities with a limited geographic focus such as these, protecting open space is motivated by amenities from recreation to simply maintaining the viewshed and 'character' of a community targeted for new urban development (Duncan & Duncan 2001; Schmidt 2008; Schmidt & Paulsen 2009).

When land conservation occurs in reaction to an imminent threat of loss, the land acquisition strategy can be characterized as reactive or "fire-fighting" (Norris & Harper 2004). That is, land subject to the greatest threat of habitat loss is prioritized for conservation. Examples of this strategy occupy a prominent place in the history and lore of land conservation because they often involve high profile, emotionally-charged campaigns showcasing grassroots activism. For example, large portions of the Yosemite Valley were secured as a National Park because of the persuasive writing of John Muir and social connections of Robert Underwood Johnson (Hall 1921). Both men were spurred into action by the extensive livestock grazing and habitat degradation they witnessed there. A more recent example of this approach to acquisition was the battle over Sterling Forest in the NY/NJ Highlands. A 1974 developer's plan to put 3,900 housing units on 1,300 acres of land in Tuxedo NY (population 3000), prompted public opposition in Tuxedo which ballooned into a twenty-five year long fight over the regional forest. The result was the formation of an wide-reaching public private partnership to combat the legal maneuverings of the development corporation and protect 20,000 acres of forest (Botshon 2007).

The complement to reactive acquisition is a proactive or "pre-emptive" strategy. That is, lands with little threat of habitat loss are secured for conservation before they become threatened (Norris & Harper 2004). Examples of this strategy are also abundant, but sometimes less celebrated in conservation history because they lack emotional charge or lack local community support, thereby shining an unflattering light on conservation. Pre-emptive acquisition was a widespread practice in the 1900s, initiated by passage of the

Weeks Act of 1911. This legislation enabled the federal government to purchase and protect the degraded landscapes left behind by intense timbering and agricultural use. These were "the lands nobody wanted" which had little appeal once, but have now regrown into over 23 million acres of valuable eastern national forests (Shands & Healy 1977). In the 1920s, John D. Rockafeller Jr., in collusion with Horace Albright, created a shell company to quietly purchase 33,000 acres of private land around scenic Jackson Hole, Wyoming before developers reached it (Righter 2000). The land was eventually donated back to the federal government to expand the Teton National Park. In the 2000s Roxanne Quimby, cofounder of the Burts Bees company, took a page from Rockafeller's playbook. With the timber industry in decline, she slowly purchased over 50,000 acres of land across northern Maine in support of a new National Park proposed there (Harrison 2006). Both Rockafeller and Quimby faced intense public opposition. Locals were furious that land use in their region should be dictated by the power of one wealthy person and the federal government. Rockafeller's donation eventually occurred, but Quimby's remains in limbo.

Each land acquisition strategy introduces a different type of risk for resource conservation in any habitat of interest (Spring et al. 2007). The 'fire-fighting' strategy increases the risk of species and habitat isolation in protected parcels; the pre-emptive strategy increases the risk that species and resources will only be protected in the least threatened areas. Because of the increasingly important role local government and NGOs are playing in land acquisition, the complex social environment in which selections are made, and because agencies may operate somewhat independently of one another or a regional conservation plan, it is important to assess how individual parcel selections aggregate into regional patterns for open space networks (Meyer et al. 2015). Analyzing the aggregate land acquisition strategies that emerge from independent actions can highlight unintended consequences for ecosystem and biodiversity protection.

My research goals are to analyze 1) how often lands of high and low conservation value transition into either protected or urban states, 2) if it is possible to characterize the aggregate regional strategy for land conservation as proactive or reactive, and 3) which environmental and social factors predict the amount of high value conservation land protected. Specifically, the following analysis examines how the relative threat of urban conversion interacts with the amount of land protected to shape regional land acquisition strategies. If lands least threatened by urban conversion show higher rates of protection, this would provide evidence that a proactive strategy best characterizes the collective actions of land acquisition agencies. If lands highly threatened by urban conversion show higher rates of protection, this would provide evidence that a reactive strategy is a more accurate characterization of the aggregate actions by regional conservation groups.

Methods

My study focuses on the New Jersey Highlands, an urbanizing ecoregion in the northern portion of the state. To evaluate the dominant land protection strategy from 2000-2012, I assembled data from spatial and non-spatial datasets and ran two analyses. My first analysis used only spatial datasets. The first was my vector layer of Protected Area with boundaries aligned to the statewide tax-parcel layer (Chapter 2). For this analysis, open space which is legally protected I will refer to as "Protected Areas" to highlight the land use. This does not imply that all the land I analyze has formal IUCN recognition as a Protected Area (Dudley 2008), but portions of it do.

The second dataset was a raster layer with a 30m x 30m grid cell resolution of all undeveloped land potentially "available" to be developed or set aside for open space as of

2000 (Phelps & Hoppe 2002). This data product was developed through a multi-agency collaboration as part of a 2002 report on the NY/NJ Highlands. Referred to as the "Conservation Value Assessment" (CVA) layer within the report, it exclude all urban and Protected Areas know by 2000 as these lands were considered "unavailable" to transition into a different land use. A brief description of the CVA attributes follows. Greater detail on its development is available in Phelps and Hoppe (2002).

The CVA layer models both the conservation value and the likelihood of conversion to urban land use for each grid cell. The conservation value of each cell was ranked using a composite score of five different resource value parameters: water quality, forest quality, biodiversity, agricultural quality, and recreation value. Composite conservation values were represented on a scale of one to five (five being the highest value). This ranking was then mapped as a binary raster in which each grid cell was assigned either "high conservation value" (for composite values of three or above), and "low conservation value" otherwise.

The likelihood that a grid cell would convert to urban land cover was derived from the relationship between fourteen econometrics parameters (Appendix C) and the urban land conversion patterns between 1995 and 2000. Like the conservation values, the overall likelihood of conversion was first represented on a scale of one through five. This ranking was mapped as a binary raster in which each grid cell was assigned either "high likelihood value" (for composite values of three or above), and "low likelihood value" otherwise.

The binary conservation and likelihood layers were combined within the CVA layer. The layer assigns all available land in the NJ Highlands to one of four classifications: 1. low conversion likelihood, low conservation value (28,221 hectares (ha)), 2. high conversion likelihood, low conservation value (21,806 ha), 3. low conversion likelihood, high conservation value (73,957 ha), and 4. high conversion likelihood, high conservation value (29,603 ha). See Appendix D for a reproduction of the CVA layer.

Transition probabilities from the CVA model

I calculated the probability that the available 153,587 ha of high and low value land from the CVA layer would transition into one of three states from 2000-2012: Urban, Protected Area, or Still Available. The first two states represented absorbing states because presumably land remained in these states once it transitioned. The third state represented land that remained available for future transitions. As such it was not an absorbing state.

To estimate transition probabilities, I intersected the PATX layer (Chapter 2) and CVA layer. I calculated the total area protected from 2000-2012, given the pool of land available from the CVA. This identified the amount of available land transitioning into the “Protected Area” (PA) state. I used the 2012 land use/land cover layer (NJDEP 2012), to select urban development and performed a similar intersection to identify the amount of available lands transitioning into the “Urban” state. I calculated the amount of land in the “Still Available” state by summing lands which were neither PA, nor Urban. For this analysis, I considered all agricultural land part of the pool of available land which could still either transition into urban or protection later.

I used the classifications from the CVA layer to assess the prevalence of proactive versus reactive land acquisition strategies. This analysis specifically compared the relative transition probabilities from the high and low likelihood categories for all land that transitioned into protection (PA). The likelihood classification provides a relative measure of the threat of urban conversion. Therefore, if I found higher transition probabilities into protection for the low likelihood groups, that would provide evidence for a proactive strategy of protection. (In other words, available land had a higher probability of transitioning into the PA state when there was no immediate threat of development.) Conversely if I found higher transition probabilities for the high likelihood groups, that result would provide evidence for a reactive strategy. (In other words, available land had a higher probability of transitioning into the PA state under a greater threat of development.)

For this analysis, finding evidence for proactive versus reactive strategies depended upon the accuracy of the high/low likelihood classifications in the CVA layer. If the CVA predictions were poor, then development pressure is not accurately represented and competing strategies cannot be assessed. Likelihood of urban conversion was predicted for each cell in 2002, but the classification accuracy can be tested against the actual urban transitions that have since occurred. Therefore, for each likelihood category, I calculated urban transition probabilities. This allowed me to assess the reliability of the original binary classifications.

I also calculated the aggregate probabilities for high and low value conservation categories (ignoring whether those lands fall into the high or low urban transition likelihood categories). These transitions are informative because they quantify how often high value lands are protected, versus developed. Conversely, this analysis of land value also highlights how often conservation actions omit high priority lands, instead favoring low-value lands. Transition patterns for high value lands are also of interest in conjunction with likelihood of urban conversion because they quantify how often high value, highly threatened lands are protected, versus developed.

Regression model for acquisition of high value conservation lands

My second analysis assessed strategies of land protection using Ordinary Least Square (OLS) regression. This modeled the correlation between six potential explanatory variables and the quantity of high value land protected in each Highlands municipality (n=83) from 2000-2012. These data were derived from both spatial and non-spatial datasets. The six explanatory variables represented environmental and socio-economic conditions in each municipality in the five to ten years preceding the study period. The source of each variable and a brief description is provided in Table 5.1.

My dependent variable was the total hectares of high quality land which transitioned into protection from 2000-2012. Using the Protected Areas layer aligned to tax parcel boundaries ("PATX" from Chapter 2), I identified the lands which transitioned into protected status after the CVA layer was produced. Because I was evaluating human behavior (i.e. proactive versus reactive strategies) I had to select variables to serve as proxies or indicators for that behavior. The two independent variables (Table 5.1) I selected as proxies for proactive land acquisition were: 1) hectares of high value land available in 2000, and 2) hectares of land protected by 2000. The four independent variables (Table 5.1) serving as proxies for reactive/fire-fighting land acquisition patterns were: 1) average income per capita, 2) amount of impervious surface added from 1995-2002, 3) population density increase from 1990-2000, and 4) the percent increase in average house price from 1995-2000. The procedure for calculating each variable and the data use is detailed in Appendix E.

Using R software (R Core Team 2016), I ran diagnostic tests for heteroskedasticity and multicollinearity prior to running the regression analysis. Breusch-Pagen test results indicated the presence of heterskedasticity which I corrected using a Box-Cox transformation on the dependent variable (Neter et al. 1996). Variable inflation factors well below recommended thresholds (Neter et al. 1996) indicated that multicollinearity was not an problem and all variables could be used. I set up six candidate models that explored model performance for just the proactive variables, just the reactive variables, and various combinations of these variables. I compared the performance of the candidate models using both R^2 values and Akaike Information Criterial (AIC) scores, following guidelines given in Burnham and Anderson (2002).

Expected relationships between dependent and proactive variables

A significant and positive relationship between the amount of high value land protected after 2000 and the first two independent variables from Table 5.1 (land available in 2000 and land protected by 2000) would provide evidence for a predominantly proactive acquisition policy. That is, most land was protected after 2000 in areas less threatened by development. Under this strategy, more high quality land would be acquired in municipalities where more high quality land was available. I also expected more land would be acquired in municipalities with less land protected by 2000 (the start of the study period). I expected a proactive strategy would show this relationship for two reasons. First, in municipalities with less protected land, more high value land could theoretically be acquired because the supply of available land would reduce competition from development, and 2) if a municipality already had a lot of protected land, there might not be much high quality land remaining (reducing the incentive to protect any remaining land).

Expected relationships between dependent and reactive variables

I used the last four explanatory variables in Table 5.1 as indicators of a reactive conservation strategy as these are metrics of development pressure within a township. A strong positive relationship between one of more of these variables and amount of land protected in a municipality would suggest that land protection occurs in response to development, but generally not before that threat emerges. My reasoning for a relationship between each metric and the dependent variable is explained below.

Wealthier municipalities may react to the threat of development by using open space preservation to restrict development (Duncan & Duncan 2001; Rudel et al. 2011). Wealthier municipalities may also have greater monetary and political capacity to support local funding for open space protection. This would produce a strong positive relationship between income per capita and the amount of high value land protected, if a reactive strategy is most prevalent.

Logan and Molotch's (2007) work on cities, and Beuschel and Rudel's (2009) work on the NJ Highlands, shows how development and land preservation can be coupled. Townships often require developers to protect land as a condition for permitting development. Therefore, a positive relationship between the increase in impervious surface and the amount of open space protected would show evidence of a reactive strategy.

Population increase can result from new development within a municipality, either through "building up" or "building out". Even in the absence of new development, desirable amenities (e.g. good schools or job opportunities) coupled with a shortage of new housing can increase population densities by encouraging subdivision of existing housing stock into smaller living units. Density increases generate more foot and car traffic, elevating the demand for open space. Previous studies have documented that communities experiencing rapid growth, are more likely to approve and fund open space initiatives (Kotchen & Powers 2006; Nelson et al. 2007). Under a reactive strategy, a strong positive relationship should exist between density increase and the amount of high quality land protected subsequently.

Housing price increases provide a relative measure of how desirable one community may be for development, relative to another. Previous studies have documented that more affluent communities are more likely to approve and fund open space initiatives (Kotchen & Powers 2006; Nelson et al. 2007). Therefore, the percent increase in housing prices prior to the study period should have a positive correlation with open space acquired in a municipalities under a reactive strategy for conservation land acquisition.

Analysis of spatial dependence in regression model

Past analyses of the Highlands have shown that socio-economic processes such as increases in open space and changes in zoning laws exhibit spatial clustering (Rudel et al. 2011). For this reason, I ran diagnostic tests using GeoDa software (Anselin & Rey 2014) to determine if my regression model exhibited spatial dependence. Tests included the LaGrange Multiplier test for spatial lag and spatial error, as well as calculation of the Moran's I statistic. Both these tests required the construction and specification of a spatial weights file, which evaluated the municipal polygons for contiguity. I selected the queen contiguity weight specification, which evaluated all neighbors touching a municipal polygon. All subsequent analyses I ran as an OLS regression within R software using the stargazer package (Hlavac 2018).

Results

CVA analysis

Figure 5.1 shows a probability tree for 'available' land in 2000 transitioning into one of three possible states between 2000 and 2012. The available land in 2000 (153,587 ha) was assigned to one of four possible groups within the CVA layer. This is shown in the diagram's first set of branches or events. The total hectares of land in each group is shown in each box below the class (group 1 "lllv" = low conversion likelihood, low conservation value (28,221 ha), group 2 "hllv" = high conversion likelihood, low conservation value (21,806 ha), group 3 "llhv" = low conversion likelihood, high conservation value (73,957 ha), group 4 "hlhv" = high conversion likelihood, high conservation value (29,603 ha). The probability of available land belonging to one of the four groups is shown just to the left of these boxes. Class 3 composed the highest relative proportion of the available land area within the CVA layer (~0.48). The other three classes each composed 14-19% of the remaining available land. All probability values sum to one is this column.

The probability of land in each of the four groups transitioning into one of three possible states is shown in the second set of branches or events in Figure 5.1. Total hectares of land from each group which transitioned into Protected Areas are shown in the top box of each branch (labeled "PA"). Total hectares of land from each group which transitioned into urban development are shown in the middle box (labeled "Urb"). Total hectares of land from each group which remained available are shown in the bottom box (labeled "Still avl"). The probability of land within a group transitioning into one of the three final states is shown just to the left of the boxes. All three probability values sum to one across the three branches extending from a single land group (e.g. lllv). The combined probability for each outcome (which is a product of the two preceding events) is shown in the rightmost column. This column sums to one.

I report first on the probabilities associated with urban transitions. Results are organized to highlight the similarity of probability outcomes among certain groups because these were unexpected and are central to interpreting subsequent results. For transitions into protected status (PA), I was most interested in comparing probabilities for low and high value conservation lands, so results are organized to facilitate that comparison first. Secondarily, I report on the low and high likelihood groups transitioning into PAs. (These are the groups with high and low threat of urban conversion.)

'Available' to Urban transition probabilities

All 'available'-to-urban transition probabilities are listed to the right of the middle set of boxes labeled "Urb" (Figure 5.1). The probability of group 1 (low likelihood low value lands) converting to urban was identical to the probability of group 4 (high likelihood, high value lands) converting to urban (0.016). The probability of group 2 (high likelihood low value lands) converting to urban was nearly identical to the probability of group 3 (low likelihood, high value lands) converting to urban (0.022, and 0.023, respectively). When summed together, the low likelihood 'available' lands identified by the CVA layer in 2000 had nearly identical rates of transition into urban as the high likelihood lands (0.039 and 0.038, respectively). These results were unexpected and suggest potential problems with the predictive power of the classifications.

When summing transition probabilities into urban land using only low and high conservation value as criteria (i.e. ignoring the likelihood category), I again found that the two group had similar probabilities (0.038 and 0.039, respectively). Thus, similar amounts of high and low value lands (5956 ha and 5766 ha, respectively) were lost to urban development. Each represented roughly 4% of total available land at the beginning of the study period.

'Available' to Protected transition probabilities

Based on the PATX layer, I estimated that 24,160 ha were protected from 2000-2012 in the Highlands (Figure 5.1). All probabilities for available land transitioning into protected status are listed to the right of the top set of boxes labeled "PA" (Figure 5.1). For group 1 (low likelihood, low value) the probability was 0.024. For group 2 (high likelihood, low value) it was 0.006. For group 3 (low likelihood, high value), it was 0.105. For group 4 (high likelihood, high value) it was 0.022. There was a notable gap between the probabilities for high and low value lands becoming Protected Areas. When summed together, the low value lands had probabilities more than four times below the high value lands. Specifically, low value lands (groups 1 and 2) were acquired with a probability of just 0.03 whereas high value conservation lands (groups 3 and 4) were acquired with a combined probability of 0.127. Thus, high value lands were more often protected than low value lands.

The total amount of land that transitioned into protection was nearly double the amount which transitioned into urban (24,160 ha versus 11,722, respectively). Of the available lands which transitioned into PA, 19,817 ha (82%) were drawn from available lands in the low-likelihood class (which I previously argued would be associated with a proactive/pre-emptive strategy), and 4,343 ha (18%) were drawn from the available lands in the high-likelihood class (which I previously argued would be associated with a reactive/fire-fighting strategy). Thus, the summed probabilities for low and high likelihood transitioning into PAs (ignoring the conservation value), were 0.129, and 0.028, respectively. However, these results can only be interpreted in the context of the urban transition results. Above, I showed that the likelihood categories were unexpectedly weak predictors of the actual urban transitions that occurred. Thus, the differences in the PA transition just cited are not meaningful for interpreting strategies because the measure of development pressure is weak. Without a reliable metric of development likelihood, the prevalence of proactive and reactive strategies cannot be assessed using these data.

Test for spatial dependence

Results from tests for spatial dependence conducted in GeoDa are shown in Table 5.2 (results shown are for the full model only from Table 5.3). The Moran's I (MI) value was low (0.1865) and did not indicate the presence of spatial clustering in the regression. Analytical outputs from both types of Lagrange Multiplier tests were also low and did not detect any patterns of spatial clustering for either spatial lag ($LM\lambda = 1.0593$) or spatial error ($LM\rho = 0.0401$). The robust versions of these tests also did not yield significant results ($LM\lambda^* = 1.6977$, and $LM\rho^* = 0.41009$, respectively).

Regression model results

Regression results for the six candidate models are given in Table 5.3. The models using only reactive variables were poor predictors for the amount of high value land protected (adjusted R^2 of > 0.20 for m3 and m4). Models incorporating the proactive variables (m1, m2, m5, m6) were strong predictors (adjusted R^2 values > 0.79). Models that used both proactive and reactive variables (m5 and m6) had slightly higher adjusted R^2 values (~ 0.83) than the ones which only used proactive variables (m1 = 0.79 and m2 = 0.81).

The sign and effect size for each variable are listed in Table 5.3 along with their standard errors (in parentheses, below each). In the four models with the highest adjusted R^2 values, both proactive variables had positive coefficients. The sign of the coefficient for the first variable (*available in 2000*) was consistent with my prediction. Having greater amounts of high value land available did increase the amount of land protected during the study. However, the sign of the second proactive variable (*protected by 2000*) was not. Having a high amount of protected land did not suppress future land protection but instead increased it.

The two reactive variables (*impervious surface added* and *population density increase)* in the top four models had negative coefficients. Thus, there was less land protected in municipalities within rising incomes and increasing amounts of impervious surface. I hypothesized that a strong positive relationship between these variables and the amount of high value land protected would provide evidence for a reactive strategy. This relationship suggests that increased development pressure does not result in more land protection and therefore the reactive strategy is not dominant across the region.

Given the similar predictive performance of m1, m2, m5, and m6, I also evaluated AIC scores. AIC values help discern differences among candidates when R^2 values are similar. The model with the lowest overall AIC score of 872.72 was the Proactive + Reactive one (m5). As the lowest, all other AIC differences for each model are calculated and interpreted relative to this model (Anderson & Burnham 2002). Differences are shown in Table 5.4. Although the adjusted R^2 only changed slightly between m1, m2 and m5, the AIC change relative to m5 was substantial (12.88 and 7.36, respectively). This was large enough to disqualify both of these models from consideration for model averaging (Anderson & Burnham 2002). The Proactive + Reactive model (m5) included both variables used in m1 and m2, plus two additional variables linked to a reactive strategy (*impervious surface added* and *income per capita*). These regression results suggest that although the proactive variables were central to predicting high value land acquisitions from 2000-2012, reactive variables also played an important role in improving the model.

The full model (m6) had an AIC score which differed from the Proactive + Reactive model by 3.32 points. Models with an AIC difference of 3-4 points (relative to the lowest AIC score) could be considered for model averaging. However, I chose not to do so because the coefficients for variables included in both models were nearly identical, and the additional two variables included in the full model were not statistically significant. Model averaging would have changed the results little and likely not in a substantive way for my research question.

Discussion

CVA analysis

The results of the analysis demonstrate that the CVA classifications were somewhat poor predictors for the actual urban land transitions that occurred during the 2000-2012 study period. If the classifications had accurately distinguished low versus high likelihood of urban conversion, the low likelihood lands would have had transition probabilities lower than the high likelihood lands. However, for the actual transitions that occurred from 2000-2012, the transition probabilities were equal (0.039 and 0.038, respectively). Shortcomings in the CVA classifications were likely the result of legislative changes rather than a poorly constructed econometric model. The CVA layer was published in 2002 as part of an effort to provide the scientific data to support protective legislation for the Highlands. That protection was legislated at both the state and federal level in 2004. This changed the standards and review process for urban development throughout the Highlands, as well as evolving the cultural conversation on the role of the Highlands in supporting human needs (Pirani et al. 2011). Consequently, it is not surprising to find changes between urban development patterns of the prior decade (on which the econometric model was based) and the 2000-2012 study period.

The weakness of the CVA likelihood predictions (for urban transitions) meant that transition probabilities from the CVA analysis could not be used as a line of evidence to assess the relative prevalence of proactive and reactive conservation strategies. Assessing strategies hinges upon properly distinguishing some degree of development pressure across the Highlands and I had no evidence that the CVA likelihood classifications did that. Therefore, I focus on the regression results as the primary line of evidence. However, results of my CVA analysis are still highly useful in one respect. They highlight the relative difference in transitions (both to urban and PA) for high value versus low value conservation lands. Because the conservation value classification is not predictive, these results can be interpreted without caveats.

From the standpoint of maximizing biodiversity and other natural resource conservation, it is preferable to have low value conservation lands transition into urban more often than high value lands. Conversely, it is also preferable that high value lands transition into protection more often than low value lands. With regard to urban transitions, my analysis showed that similar amounts (~4%) of high and low value lands were lost to urban development. This means that urban development did not distinguish or "spare" high value lands from being developed during this period. However, with regard to PA transitions, there was twice as much land protected as developed. Furthermore, high value lands were protected with a probability more than four times greater than the low value lands. These are important findings. They demonstrate favorable aggregate land selection patterns for regional land conservation networks, even though agencies can exercise autonomy in conservation land selection. Thus, multiple organizations, sometimes working independently and sometimes working in partnerships, collectively preferentially selected high value lands for protection.

These favorable choices for assembling regional conservation networks may have been facilitated by the availability and guidance of the CVA layer. Following its production in 2002, it was distributed regionally through information and training sessions targeting land acquisition agencies (R. Lathrop, pers. comm). By this point, a collaborative effort to coordinate and set regional conservation policies had coalesced after .a 1992 U.S. Forest Service report on the Highlands unlocked federal funding to support the necessary research and planning. Consequently engagement in a regional planning process for the Highlands has been high and has incorporated all levels of government and NGO administration. The 2004 NJ Highlands Water Protection and Planning Act mandates that municipalities follow development and land conservation practices compatible with the Highlands Master Plan (The Highlands Water Protection and Planning Council 2008). Since the 2004 Act primarily targets development reform, it falls short of mandating that conservation acquisitions adhere to a specific SCP. Agencies can still therefore exercise autonomy in the lands they protect, although securing matching government funds is inevitably easier for documented high value lands.

Both the 1992 and 2002 report produced data layers in which priority conservation lands were identified, each layer being more comprehensive about integrating regional conservation priorities than the last. To date, no data have been collected regarding how the CVA or other spatially explicit land conservation priority data have influenced the parcel-level decision made by land acquisition groups. This chapter also does not address this question specifically because the data span a time when the regional planning process was under development. While there are decades of data on PA acquisition before the planning process, any post-planning process data would be smaller and subject to caveats because institutional capacity for using GIS data layers would inevitably vary by institution through time.

This question of institutional capacity for accessing and following regional GIS conservation priority layers could be addressed in future work through both quantitative and qualitative analysis. If conservation parcels were all assigned a “date protected” variable using tax records, conservation choices in different time periods (pre-1992, 1992-2002, and post 2002) could be compared. Qualitative analysis could also be conducted through interviews with agencies to determine how institutional capacity for GIS use and land acquisition funding has changed over those same time steps. State agencies, for example have likely had greater capacity than NGOs since the beginning of the planning process.

OLS regression analysis

Out of all six candidate models, I selected the proactive + reactive model (m5) as the best because its components explained the most variability in the least complex model. It showed that although a proactive strategy most generally characterized the acquisitions from 2000-2012, reactive variables also played an important role in accurately modeling the amount of conservation land acquired. Thus, the conservation acquisition strategy cannot strictly be defined as either proactive or reactive. It was both, although not in equal portions.

In aggregate, the agencies involved in Highlands land acquisition heavily favored proactive acquisitions for the period under study. The largest amounts of land were protected in municipalities where more land was available and there were greater pre-existing amounts of protected land. Available land had a positive effect, consistent with my predictions. However, the response to the latter variable (existing protected land) requires more interpretation. Contrary to my expectations, pre-existing protected land did not suppress future land protection. Rather, it increased it.

Is this still evidence of a proactive strategy? Given that my analysis was restricted to only high quality land, this might not be inconsistent with a proactive strategy. I had expected that municipalities would acquire only enough land to meet specific goals for open space. For example, within their Open Space plans, many municipalities reference goals set by the National Parks and Recreation Association (NPRA) (Township of Chatham Open Space Advisory Committee 2010). This is currently ~10 acres of 'active' recreational municipal open space per 1000 people. These NPRA guidelines do not include any specific goals for 'passive' recreation (undeveloped open space). This reduces the incentive for counties and municipalities to protect large quantities of additional land once active recreation goals have been met. Therefore, I expected that acquisition would taper in one municipality, but then increase in another as development pressure shifted and demand increased (Armsworth et al. 2006; Butsic et al. 2013).

However, this line of reasoning does not consider the spatial clustering of development and its relationship with high and low quality conservation land. Development and habitat loss tend to be spatially dependent, progressing along travel corridors from metropolitan areas, proceeding from the coast inward, and moving from low to high elevations (Seabloom et al. 2016). In the Highlands, this pattern also holds. The largest amount of available high quality land was concentrated within a few municipalities with sparse development, higher in elevation, and distant from travel corridors (Phelps & Hoppe 2002). It seems likely that agencies operating outside municipal interests are still proactively investing in expanding existing conservation lands there. Consequently, the positive relationship between high amounts of protected lands by 2000, and the amount of high value land protected from 2000-2012 could arguably provide evidence for a proactive acquisition strategy.

The inclusion of reactive variables (*impervious surface added* and *population density increase*) in the Proactive + Reactive model are important to note because they improve the model even through their effect size is small. Their inclusion suggests that land protection does not compete well in municipalities with desirable commercial and residential land markets. When land becomes valuable for development it suppresses the total amount of land acquired for protection. This does not necessarily mean that there is less money spent on conservation in these communities. In fact, fast growing, more affluent communities in New Jersey and elsewhere in the country approve and spend significant amounts of money for open space protection (Kotchen & Powers 2006; Nelson et al. 2007). If conservation spending per municipality were the response variable, reactive and proactive acquisition strategies might show more balance.

Although this analysis shows strong evidence for the prevalence of a proactive strategy across the Highlands, it does not test hypotheses for drivers of this behavior. Such behavior could be explained by the cost of land (Armsworth et al. 2006; Kim et al. 2014; Cho et al. 2017). Land valuable for development will command a higher price and reduce the total hectares that can be conserved. New Jersey state government has shown a preference for acquiring larger, less expensive tracts as a means to stretch taxpayer dollars and shine a favorable light on the open space program (Foresta 1981). Indeed, one of the most contentious purchases in New Jersey Green Acres history was that of the urban land for Liberty State Park because the cost per acre was so high (Foresta 1981). Other organizations likely experience similar pressure to stretch funds. Future studies in this area would benefit by testing drivers for the acquisition strategies reported here.

Conclusions

Spring et al. (2007) suggests that a balance between proactive and reactive strategies is the most robust approach to biodiversity conservation, given uncertainty about the timing of urban development, cost of land and the shape of species-area curves for protected parcels. All these uncertainties coexist in urbanizing regions, so a predominantly proactive approach, such as the one documented in the New Jersey Highlands, is arguably not optimal. However, if conservation spending per municipality were the response variable, reactive and proactive acquisitions might align better with a robust strategy. Spending data were not part of this study, but would be worth incorporating into future research for a different perspective on this question. Reactive strategies perhaps should not only be defined through the amount of land protected, but also the amount of money government and citizens are willing to contribute at the most critical times.

My analysis has implications for other urbanizing regions that have shown strong support for local open space funding initiatives (e.g. Florida, California and Massachusetts (Lerner et al. 2007; Nelson et al. 2007). Small land acquisitions spurred only by the imminent threat of development often feel like “too little too late” when compared with the formal practice of Systematic Conservation Planning (Margules 2000; Sarkar et al. 2006). Such last-minute acquisitions may also seem like an inefficient use of conservation funds when land prices are high. However, the conservation work done through more reactive local ballot measures may play an important role in protecting the most threatened land. These lands might otherwise be ignored by larger government and NGO’s focused on satisfying contributors that spending is being done efficiently.

Table 5.1 The variables collected for each Highlands municipality (n = 83) used in the regression analysis. The source for the data and a description of each variable is provided below.

Protected after 2000	The total hectares of high value land which transitioned into protection from 2000-2012. Source: The CVA layer (Figure 3-20 in Phelps and Hoppe 2002), and the PATX data (see Chapter 2: Methods)
Available in 20000	The total hectares of high value conservation land "available" by 2000 for either protection or development. Source: NJDEP (2014) and the CVA layer (Figure 3.20 in Phelps and Hoppe 2002)
Protected by 2000	The total hectares of land protected by 2000. Source: PATX data (see Chapter 2: Methods)
Impervious surface added	The hectares of impervious surface added from 1995-2002 in each township. Source: NJDEP (2007)
Income	The average income per capita in 2000, reported in U.S. dollars. Source: U.S. Census Bureau (2000)
Population density increase	The population density increase from 1990-2000. Source: NJDEP (2014) which reports population densities by municipality according to the 1990 and the 2000 U.S. Census.
House price increase	The percent increase in mean housing price from 1995-2000. Source: New Jersey Department of the Treasury (1995, 2000)

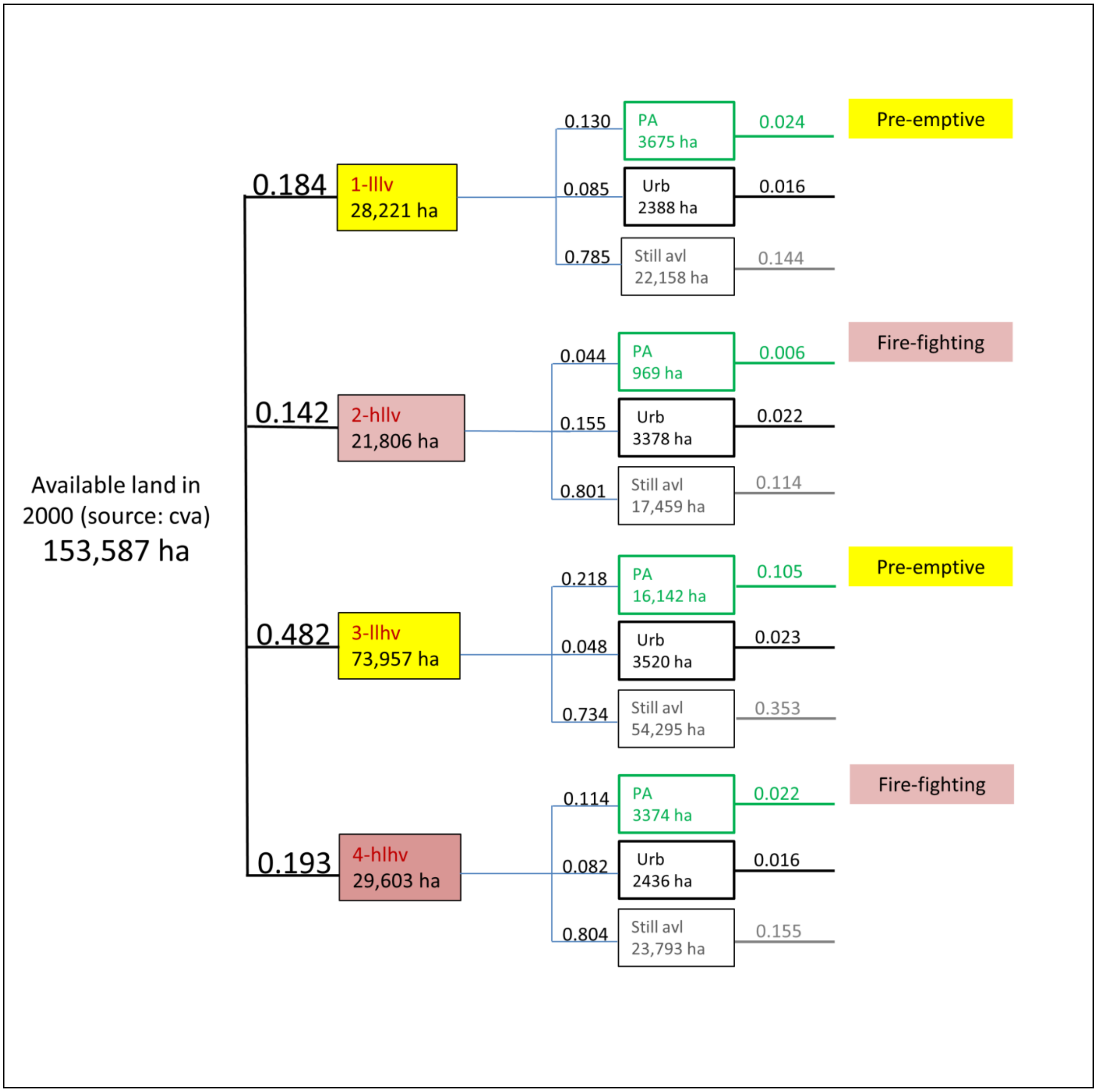


Figure 5.1 The diagram shows the probability of available land belonging to one of four possible groups in 2000, and entering one of three possible states by 2012. Probabilities for each individual event are shown to the left of the boxes. Total probabilities for each possible outcome (i.e. four group types entering one of three possible states) are calculated in the final right hand column.

(Key for land group types: 1 lllv = low conversion likelihood, low conservation value, 2 hllv = high conversion likelihood, low conservation value, 3 llhv = low conversion likelihood, high conservation value, 4 hlhv = high conversion likelihood, high conservation value. Key for land use states by 2000: PA = Protected Area, Urb = Urban land, Still avl = indicates land did not transition into either PA or Urban states and is therefore still available for future transitions.

Table 5.2 Diagnostic test results for spatial dependence among the 83 Highlands municipalities are shown. Tests were performed in GeoDa software using a queen contiguity weight matrix with row-standardized weights.

TEST	NOTATION	MI/DF	VALUE	PROBABILITY
Moran's I (error)		0.0156	0.1865	0.85209
Lagrange Multiplier (lag)	LM_λ	1	1.0593	0.30339
Robust LM (lag)	LM_λ^*	1	1.6977	0.19259
Lagrange Multiplier (error)	LM_ρ	1	0.0401	0.84132
Robust LM (error)	LM_ρ^*	1	0.6785	0.41009
Lagrange Multiplier (SARMA)	$LM_{\rho\lambda}$	2	1.7378	0.41941

Table 5.3 Six candidate models were selected to test the variables associated with proactive and reactive land acquisition strategies. Models using only reactive variables were poor predictors for the amount of high value land protected. The proactive + reactive model (m5) had the lowest AIC score and a high adjusted R^2 value.

The variables (rows) in each of the six candidate models (columns) are shown below with model statistics. Coefficients are reported in the top portion of the table with standard errors shown below them in parentheses. Summary statistics are reported in the bottom portion of the table.

	Dependent variable: High value land protected after 2000					
	Proactive1 (m1)	**Proactive2 (m2)**	**Reactive1 (m3)**	**Reactive2 (m4)**	**Proactive + Reactive (m5)**	**Full Model (m6)**
Available in 2000	0.070*** (0.004)	0.063*** (0.005)			0.067*** (0.005)	0.067*** (0.005)
Protected by 2000		0.014*** (0.005)			0.015** (0.005)	0.015*** (0.005)
Impervious surface added			1.878*** (0.518)	1.821*** (0.522)	-0.470*** (0.265)	-0.500* (0.297)
Income per capita			-0.001 (0.001)	-0.001 (0.001)	-0.001*** (0.0004)	-0.001*** (0.0005)
Pop. density increase				-19.160* (11.092)		-0.989 (5.112)
House price increase				0.196 (0.871)		0.312 (0.393)
Constant	-11.985 (7.315)	-12.255* (7.035)	77.786** (31.915)	93.007*** (33.394)	26.682* (14.473)	29.040* (15.541)
AIC	885.60	880.08	1005.76	1006.76	872.72	876.04
Observations	83	83	83	83	83	83
R^2	0.795	0.813	0.149	0.180	0.837	0.838
Adjusted R^2	0.792	0.808	0.127	0.138	0.828	0.825
Residual	49.013	47.133	100.494	99.848	44.578	44.974
F Statistic	313.983***	173.563***	6.977***	4.294***	99.871***	65.518***
Note:				*p<0.1; **p<0.05; ***p<0.01		

Table 5.4 The calculated differences in AIC scores are show below, using m5 as the best model and standard for comparison.

Models compared	AIC difference
m1-m5	12.88
m2-m5	7.36
m3-m5	133.04
m4-m5	134.04
m6-m5	3.32

Chapter 6: Conclusions

As a consequence of changes made in federal open space funding policies and increased support from citizens and special interest groups for local open space funding, the selection of land parcels for conservation has become a more democratic and decentralized practice in recent decades. This is particularly evident in certain urbanizing areas where open space taxes have been approved at the both the county and municipal level. One theoretical question emerging from this shift is: Has this approach helped build PA networks with favorable spatial characteristics for resource conservation, or is this a failed experiment that should be avoided? More involvement from small, specialized public and private land trusts has great potential to increase access to acquisitions opportunities and funding, thus protecting more land overall. However, the proliferation of conservation groups with specialized goals, restricted funding, and biased geographic focus could also yield a network of protected habitat with small, isolated parcels of poor conservation quality. My work takes a spatial approach to the question, trying to index 'success' or 'failure' by comparing patterns of conservation lands in the Highlands to some basic spatial guidelines for land conservation networks.

My results suggest that the emerging NJ Highlands PA network has several spatial patterns favored in the design of land conservation networks. The parcelization study (Chapter 3), showed that the largest forest fragments in the region often had 80% or more of the forest protected. High percentages of protection in habitat patches are preferable to having only a portion protected because it can limit future fragmentation of that habitat patch. The analysis of Protected Forest boundary changes (Chapter 4) showed that some of the smaller PFs gained natural habitat along their boundaries. Particularly for smaller protected habitats, adding natural habitat at the boundary can increase the ability of species to use and traverse unprotected land. A proactive acquisition strategy (Chapter 5) dominated acquisition patterns from 2000-2012. The strategy highlights the spatial bias that is occurring in land acquisition. High amounts of high quality land are not preferentially acquired in the areas with greatest development pressure. Ecological communities in those areas are more vulnerable to loss at present. These results suggest that there are ecological benefits for the regional PA network realized by having many agencies identify and respond to land acquisition opportunities.

However, my results also reflect some potential costs associated with having so many actors selecting conservation parcels. There were nearly 8000 Protected Forest patches under five hectares which I excluded from the study because they were isolated from other patches and their function and quality was unclear. Overall, they composed ~6% of the total PF area, and less of the PA area. If they are all true PAs (and not just data entry errors), their large number and small size suggest a considerable amount of time and funding may have been spent on land which may have little conservation value. I lack data on the specific purchase costs and reason for their inclusion in PA dataset. However, they raise questions about how funding resources and tax breaks could have been better allocated to enhance the PA network.

The parcelization analysis (Chapter 3) also highlights some potential risks of future PF habitat degradation introduced by the current acquisition system. The 100-500 ha class of forest fragments represented about a third of the total existing forest area, but more than half the individual forest fragment had less than 50% of their land protected. This shows that a piecemeal approach to habitat conservation is common. This might be a symptom of the constraint on available land, or of the fact that agencies compete for funding and therefore may make smaller and more palatable acquisition requests. Systematically protecting all portions of given fragment in this size class before acquisitions begin in another fragment would be a preferable approach that could reduce future risk of forest fragmentation around protected forests. One specific inefficiency my results revealed was the acquisition of about 4600 ha of land with low conservation value (Chapter 5). This was a relatively small percent (3%) of the total available land, but again suggested that scarce funding may have been used ineffectively.

Overall, my analysis suggests that this social experiment in engaging a diverse and diffuse collection of actors in PA land acquisition has likely had more positive than negative implications for the future ecological integrity of Highlands PA networks. I base this conclusion primarily on the large amount of high quality land protected, and the fact that so many large forest fragments have been fully protected. Of course any conclusion is subject to caveats. Decentralizing the selection process for conservation lands may still not a preferable approach to building a PA network everywhere. Positive outcomes within the NJ Highlands were likely enhanced by the regional Highlands planning process that started back in 1992. Much of the PA data used in Chapters 3 and 4 pre-date this planning process, so these chapters offers the most potential insight into how conservation procedes in the absence of coordinated regional planning. However, to fully develop that picture, more work would need to be done on the PA dataset to assign year of acquisition to conservation parcels. This would enable a "before-and-after" comparison of acquisition patterns, relative to introduction of regional planning tools and practices Some temporal acquisition data were collected for earlier studies (Beuschel & Rudel 2009; Rudel et al. 2011; Gottlieb et al. 2012) and have been integrated into the dataset, but many conservation parcels lack this information, or have only a range of dates associated with it (i.e. pre-1975), so efforts to supplement the existing data are still needed to effectively address this question

The Highlands regional planning process has produced several GIS layers (referenced earlier) with parcel-specific conservation priorities which currently serve as the default SCP for PA network acquisition that all public and private conservation groups could follow. Following recommendations from these layers would maximize the conservation returns from every acquisition. However, the static nature of any GIS layer imposes limitations. Conservation priorities in the layers are based on patch-specific attributes that do not get updated according to changes in landscape context. Work is currently underway which recognizes the importance of land change around priority habitat (The Nature Conservancy et al. 2017). This is a web tool designed to make acquisition priorities as dynamic as the landscapes surrounding them. The introduction of such tools offers new opportunity to investigate how institutional capacity and technology interact to shape emerging PA networks.

One impediment to adoption of this planning tool may be that the engagement in locally-focused land acquisition, currently widespread in New Jersey, is arguably often motivated by the desire to defend one's own back yard rather than optimizing ecological conservation in a regional network. This may explain why over 4000 ha of low priority land received protection during the study period. However, even if the land acquisition is a 'defensive' action motivated by personal benefit, there is reason to believe that these actions can aggregate into meaningful reforms for building future Protected Area networks (Rudel 2013). Thus, the greatest benefit of this experiment in decentralizing PA land selection is perhaps one that has not yet been realized. The most important function of engaging so many diverse actors may not be building the optimal PA network possible today. Some choices will always be sub-optimal as long as there are no mandates for land acquisition groups to adhere to recommendations. Rather, the most important function of engaging so many diverse actors may be weaving the social fabric to optimally manage and adapt that network in the future because more stakeholder's interests have be represented. In other words, even the low quality land may serve a purpose by engaging those actors who selected it in more wide-reaching conservation agendas. With that, I believe I have waded just far enough into social theory to show the full extent of my ignorance. I will leave that thought to simmer with the hope that it will yield something fruitful 30 years from now.

Appendix A

A comparison of forest fragment (FF) count, summed total area, and relative percent of different size classes. Values for upland forests are compared to values obtained when upland and wetland forests are treated as the same land cover type. This comparison shows that distributions in size classes change slightly if wetland forest cover is included as "Forest". (All numbers have been rounded to the nearest value.)

Size class (in ha)	Upland FF count	Upland + wetland FF count:	Upland FF ha	Upland + wetland FF ha	% of total upland FF area	% of total upland + wetland FF area
0-1	11,806	8369	4042	2889	2.6	1.6
1-5	4485	3427	9733	7497	6.3	4.1
5-10	927	827	6562	5853	4.2	3.2
10-25	749	758	11,761	12,131	7.6	6.6
25-50	313	368	11,058	13,214	7.1	7.2
50-100	243	303	16,966	21,481	10.9	11.7
100-500	264	326	52,867	67,388	34.1	36.8
500-1000	32	32	21,156	21,862	13.7	11.9
1000-5000	13	17	20,848	30,990	13.5	16.9
Total	18,832	14,427	154,994	183,305	100.0	100.0

Size class (in ha)	Upland FF count	Upland + wetland FF count:	Upland FF ha	Upland + wetland FF ha	% of total upland FF area	% of total upland + wetland FF area
0-1	6128	4264	1441	992	1.8	1
1-5	1776	1477	14,020	3405	5.1	3.6
5-10	460	422	3300	2977	4.2	3.2
10-25	397	466	6153	7382	7.8	7.9
25-50	179	202	6143	7082	7.8	7.6
50-100	121	145	8521	10,257	10.8	11.0
100-500	106	119	24,907	25,961	31.5	27.7
500-1000	17	30	12,903	14,294	16.3	15.3
1000-5000	7	13	11,600	21,254	14.7	22.7
Total	9191	7138	88,988	93,604	100.0	100.0

Appendix B

The buffer mosaic transitions are listed below according to the PF size class (shown in upper left-most cell of each table). The class from which a buffer transitioned in 1986 is shown in the first column. The class into which it transitioned in 2012 is read from the second row. Values along the diagonal (in grey text) show the count of PF buffers with no transition.

5-10 ha	2012						
1986	A	an_	adn	D	dn_	N	Total
A	24	1	10		1		36
an_		15	22		1	15	53
adn	2		56	5	9	10	82
D				27	2		29
dn_			1	11	30	3	45
N			10	4	27	174	214
Total	26	16	99	47	70	202	460

10-25 ha	2012						
1986	A	an_	adn	D	dn_	N	Total
A	15		5				20
an_		6	18			5	29
adn		2	53	1	10	6	72
D				16	4		20
dn_				2	24	3	29
N		2	5	2	16	202	227
Total	15	10	81	21	54	216	397

25-50 ha	2012						
1986	A	an_	adn	D	dn_	N	Total
A	2	1	5			1	9
an_		9	1			1	11
adn			30	1	6	3	40
D				3			3
dn_				2	8		10
N			5		9	92	106
	2	10	41	6	23	97	179

50-100 ha	2012						
1986	A	an_	adn	D	dn_	N	Total
A							
an_		3	1			1	5
adn		1	18		1	3	23
D							
dn_				1	6	1	8
N			4	1	7	73	85
Total		4	23	2	14	78	121

100-500 ha	2012						
1986	A	an_	adn	D	dn_	N	Total
A							
an_			1			1	2
adn			6		2		8
D				1			1
dn_					2		2
N			1		4	88	93
Total			8	1	8	89	106

500-1000 ha	2012						
1986	A	an_	adn	D	dn_	N	Total
A							
an_							
adn						1	1
D							
dn_							
N					1	15	16
Total					1	16	17

1000-5000 ha	2012						
1986	A	an_	adn	D	dn_	N	Total
A							
an_							
adn							
D							
dn_							
N						7	7
Total						7	7

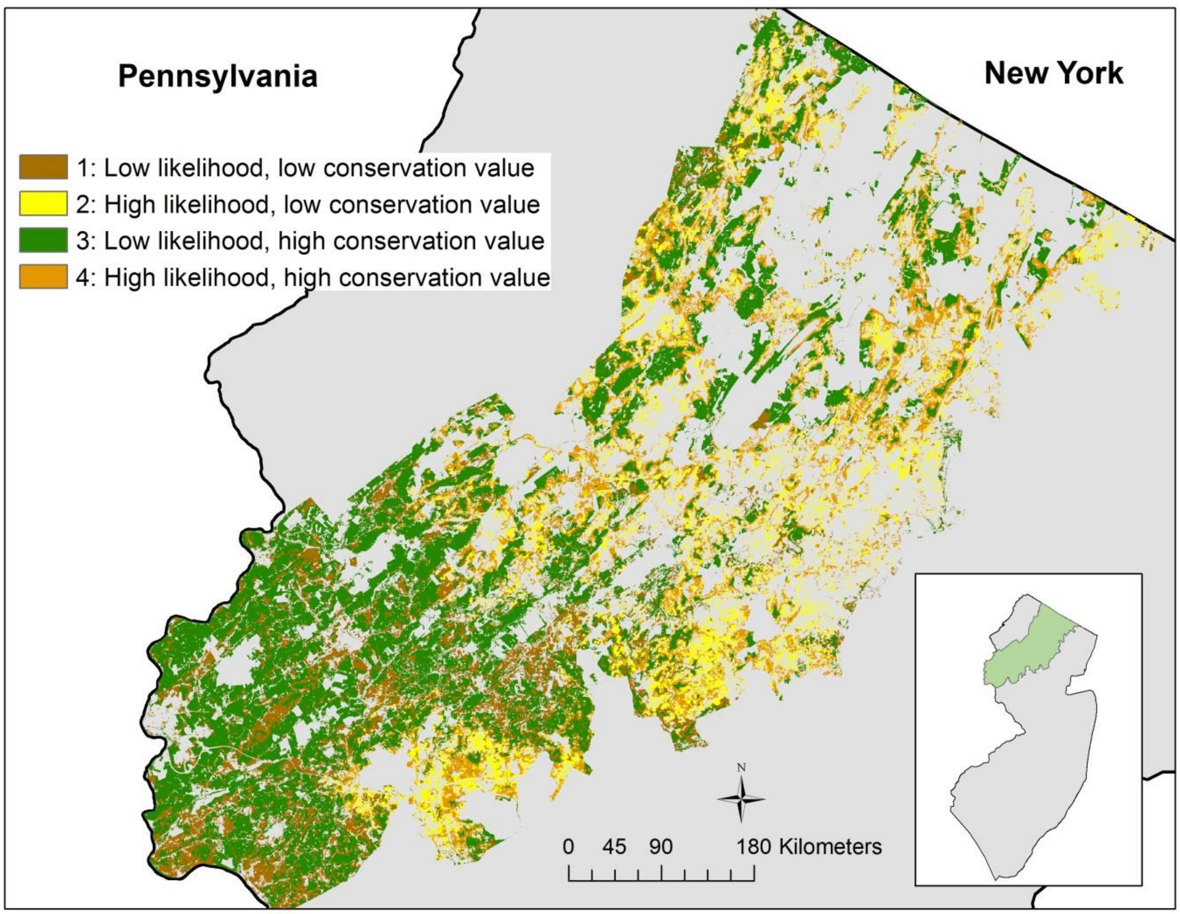
Appendix C

The fourteen parameters used in the econometric model to estimate the likelihood of conversion to urban land cover (Phelps & Hoppe 2002).

1. Distance to nearest existing developed lands
2. Participation in the Forest Stewardship Program
3. Floodprone areas
4. Prime farmland soils
5. Slope (angle of terrain)
6. Distance to the nearest water body
7. Census measures of population density (by block group)
8. Census measures of housing density (by block group)
9. Census estimates of home value (by block group)
10. Travel distance to employment centers
11. Travel distance to train stations
12. Travel distance to New York City
13. Zoning type (e.g., residential, commercial, industrial)
14. Zoning density (based on minimum lot sizes)

Appendix D

The figure below is a reproduction of the New Jersey portion from Figure 3.20 in Phelps and Hoppe (2002). Grid cells were ranked on a scale of 1-5 for their overall conservation values and likelihood of conversion to urban land cover. Conservation value maps and likelihood of conversion maps were converted into binary maps to identify lands with low and high values for each theme. Combining those binary maps yielded the four classifications shown above.



Appendix E

Calculation of Variables

Dependent Variable

High value land protected after 2000 - the total high value hectares of land which transitioned into protection from 2000-2011. Source: All acquisitions occurring in 2000 or later from the PATX data (see Chapter 3: Methods). I intersected the CVA layer with the PATX layer. Areas where the two did not overlap were selected into a new layer as post-2000 acquisitions.

Independent Variables

Proactive variables

High value land available in 2000 - To calculate the available high value land, I vectorized the CVA layer, and intersected that layer with a layer containing New Jersey municipal boundaries (NJDEP 2014). (Tabulate Intersection tool, ArcGIS 10.2). This yielded a spatial data layer of "available" land within each municipality in 2000, from which I selected only the high value lands (according to the CVA layer). Because the PATX layer (see Chapter 2: Methods) showed that some land protected by 1999 were included in the CVA layer (in other words, the CVA layer showed them as being available in 1999), I removed these protected polygons from this dataset. The final data product retained the conservation classifications (e.g. high conservation value, low likelihood), for the probability calculations.

Protected by 2000 - To calculate the land protected by 2000, I selected all PAs established prior to 2000 from the PATX data. PATX had more acquisitions occurring prior to 2000 than the CVA layer. In other words, my compilation of PA records indicated that some PAs were established by 2000, but were omitted in the CVA data. These were likely more recent acquisitions which had not yet been updated within the datasets available at the time the CVA layer was built. I included these in this variable.

Reactive variables

Income per capita – I obtained the data on mean income per capita in each township directly from the 2000 census database (U.S. Census Bureau (2000).

Impervious surface added – I calculated the hectares of impervious surface (IS) added from 1995-2002 from the 2002 land use/land cover data NJDEP (2007). The layer includes columns for both IS in 1995 and IS in 2002. Although there are impervious surface estimates available within the CVA layer, it is a raster layer produced using different methodologies than the land use/land cover data (a vector layer). Because of this, calculations done using the two vector layers yielded more accurate IS increase estimates. Consequently, the IS data go slightly beyond the year 2000, so these estimates are not restricted to the five years preceding the 2000-2012 acquisitions data I analyze. However, given that the acquisition period under analysis is fairly wide (10-12 years), allowing the IS data to overlap the acquisition data slightly seemed unlikely to alter the overall results.

Population density increase – The population density increase from 1990-2000. I calculated from U.S. Census data for 1990 and 2000 which is included as attributes within the dataset of New Jersey Municipalities (NJDEP 2014). Values are reported as increase in number of people per hectare in a municipality (Pop2000 – POP1990/ muniha).

Percent increase in house price – I calculated the percent increase in average home price from two data tables available from the State of New Jersey Department of the Treasury (Data Tables: 1995 Average Residential Sales Price, 2000 Average Residential Sales Price). Values are given as the percent increase in U.S. dollars.

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