Effects of land-use policy, forest fragmentation, and residential parcel size on land-cover and carbon storage in Southeastern Michigan

by

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A dissertation submitted in partial fulfillment
of the requirements for the degree of
Doctor of Philosophy
(Natural Resources and Environment)
in The University of Michigan
2009

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To my family, friends, and colleagues.

ACKNOWLEDGEMENTS

Sometime ago I came to the realization that there are few "self-made" people in this world. When I think about the seemingly infinite number of social interactions I've had, from childhood until writing this dissertation, it seems that it would have been impossible to predict this wonderful outcome. Even more overwhelming is the appreciation I have for those I have learned from, shared life with, and who have guided me to this point in my life. Thank you.

The Joe Sackler Colloquium in Irvine California was the first conference I was to attend as an aspiring academic. At this conference I had the great fortune of meeting Dr. Daniel Brown, who would shortly after become my Ph.D. advisor. While testing and pushing me to my limit, he has provided me with every possible opportunity I could desire during my degree. Dr. Brown has been the ultimate advisor and mentor. It's been an extraordinary amount of work, but it has been just as much fun. Thank you.

In relation to the chapters within this dissertation, I ask the reader to see the acknowledgements following those chapters for reference to those who facilitated the work within. However, I would like to formally thank my committee members Dr. Rick Riolo, Dr. William Currie, and Dr. Scott Page. Each of you has significantly assisted my journey to completion. I would also like to thank my co-authors. Dr. Brown coauthored Chapter Two with me, which is now published in the Journal of Geographical Information Science. Chapter Three was coauthored by Dr. Brown and Dr. Currie, which has been accepted for publication to Ecological Modelling. I would like to note that Dr. Currie played a significant role during the completion of my Ph.D., for it was he who suggested looking into the effects of forest patch edge on ecosystem functions. Thank you.

In relation to my personal sanity while accomplishing this dissertation, I have to thank my family and friends. Both have been crucial aids to me through the difficult and challenging periods as well as helped to celebrate the successes and opportunities that have come along the way. Thank you.

Lastly, let me give thanks to others who should also receive recognition. My apologies to those not listed that should have been. Thank you all very much.

The SLUCE group –Dr. Moira Zellner, Dr. William Rand, Dr. Joan Nassauer; Dr. Arun Agrawal; The Robinson, Mores, and Whitaker families; Becca Sowder and the Sowder and ZalZal families; Dr. Peter Deadman; Pete and Meaghan Johnson; Mike and Dominique Norton; Jon and Robyn Orazietti; Adam and Shannon Kasper; Pete Dudek; John and Sara Hill; Greg Jacobs; Courtney Murdock; Erik and Angela Dayrell; Neil Carter; Jenna Jones; my office mates, 'the great' Li An, Jason Taylor, TingTing Zhao, Geoffrey Duh, Amy Burnicki, Qing Tian, Kyle and Zana Kwaiser, Jun Wang; Shannon Brines; Bill Welsh; Andrew Bell; Greg Sawicki; the Center for the Study of Complex Systems, especially Howard Oishi; Diana Woodworth and the SNRE OAP; The Graham Environmental Sustainability Institute and its Fellows; Marco Huigen; Darla Munroe; Dawn Parker; Marco Janssen; Jan McAlpine; the land-use science community; Hugh Stimson; Jennifer Dowdell; and all other family, friends, and colleagues who have shared their thoughts and ideas along the way. Thank you.

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ABSTRACT

The overarching goal of this dissertation is to improve our understanding of the coupled natural-human land-use system in Southeastern Michigan. To accomplish this task Chapter Two presents an implementation of the DEED (Dynamic Exurban Ecological Development) model, which was used to evaluate the effects of land-use policies on forest cover. This research demonstrates one way to improve our understanding of how policy and land-use and land-cover change (LUCC) interact and can influence aggregate forest cover. The chapter provides novel contributions in the form of a framework for evaluating land-use policy effects on development and land cover, an approach to integrate an agent-based model with a geographic information system (GIS), and new examples of methods to empirically inform agent-based models.

To extend coupled natural-human systems research to include the ecological effects of LUCC and policy scenarios, Chapter Three presents an analysis of the effects of forest patch size and shape, and landscape pattern, on carbon storage estimated by BIOME-BGC. New insights from this research showed 1) the inclusion of within-forest-patch air-temperature heterogeneity can significantly influence carbon storage estimates, 2) that carbon storage estimates increase logarithmically with increasing forest fragmentation when only within-patch heterogeneity of air temperature is considered, and 3) the utility of integrating GIS and BIOME-BGC for site data collection and visualization of results.

To better evaluate the effects of land-use development policies on land-cover change and ecosystem function, effectively combining the products of Chapters Two and Three, an analysis of land cover at the residential parcel level was necessary. Land-cover analyses at the parcel level have rarely been done. Chapter Four takes a step to remedy this problem by presenting new data that describe the quantity, fragmentation, and autocorrelation of land-cover within residential land-use parcels. Results from Chapter Four could extend the policy analysis in Chapter Two, conducted at the subdivision level, to the individual parcel such that we could evaluate policies that affect individual residents and their behavior. By capturing the distribution and patterns of land-cover types across different parcel sizes we can begin to understand the linkages between household land-cover behaviors, neighborhood interactions, and landscape patterns.

Chapter 1 INTRODUCTION

1. Background

Land use and land cover change (LUCC) research has a range of themes and meanings for scientists depending on their disciplinary origin and preferred tools of analysis. For example, a geographer may use remote sensing to answer how much agriculture has been developed between 1970 and 1990 (e.g. Brown 2003), a landscape architect may be interested in how the aesthetics of the landscape influence residential settlement patterns and development (e.g. Nassauer 1992, 1995), and an economist may be interested in how land-market prices change with LUCC. What is common from each of these perspectives is that they describe 1) how humans use the land (i.e. land use), 2) what is on the land (i.e. land cover), 3) changes in land use and land cover over time, and, perhaps most importantly, 4) what processes drive LUCC across space and through time.

While geographers, such as von Thünen, Christaller, Burgess and Hoyt, and others provided early research into LUCC as it pertained to urban growth, prominence as a research topic increased when scholars recognized that LUCC was the second largest contributor to global climate change, second only to the burning of fossil fuels. Globally, land-use change between 1980 and 1989 contributed 1.6±1.0 Pg of C yr^{-1} to the atmosphere (1 pg = 10^{15}g = 1Gt) and represented approximately 30% of anthropogenic efflux of carbon to the atmosphere (Dixon et al. 1994). Despite a formal recognition of the importance of LUCC on a global scale that could affect all humans (Houghton et al. 1999, Schimel et al. 2000, Barford et al. 2001), perhaps what is most interesting and intriguing to scientists is the fact that there is no single driver of LUCC and that the global effects and consequences emerge from the bottom-up through a number of local complexities and feedbacks (Foley et al. 2005).

What this means is that people in rural and urban areas, and those on the fringe between the two, were and are contributing to a significant alteration of the earth's surface. These surface changes affect the albedo (Pielke Sr. et al. 2002); sensible and latent heat flux, evaporation (Betts et al. 1996); biodiversity (Poschlod et al. 2005);

biophysical characteristics that contribute to nutrient and hydrological cycling (Hubacek and Vazquez 2002); and carbon (C) storage (Dixon et al. 1994); each of these factors significantly influences global climate (Riebsame et al. 1994).

In contrast to the identification of LUCC solely as a source of C efflux to the atmosphere, some researchers identified a net carbon sink due to forest expansion in the 1990's in the mid-to-high latitudes (Fan et al. 1998, Casperson et al. 2000, Gower 2003). The non-stationarity of LUCC processes contributing to and mitigating climate change across space and time, and the necessity of governments to formally recognize climate change and commit to carbon budgeting and initiatives designed to ameliorate climate-change, has placed emphasis on the importance of studying land-use and land-cover change or as some refer to it, land-change science (Rindfuss et al. 2004).

This introductory chapter describes three research endeavors designed to advance the field of land-use science and improve our understanding of the bottom-up processes that drive LUCC and aggregate to influence global climate. The second chapter describes the integration of a commercial geographic information system with an agent-based model to evaluate the effects of land-use policy on forest cover in Scio Township, Southeastern Michigan. The integrated model is used to answer the following question:

> Do policies directed at private land-owners (i.e. minimum lot-size zoning) produce more aggregate forest cover than public land acquisitions made for forest conservation?

The third chapter of this dissertation identifies a gap in the representation of ecosystem function in the LUCC model presented in Chapter Two and more generally in the land-change science community. This challenge is addressed by describing results from modeling carbon storage in the highly fragmented and human-dominated environment of Southeastern Michigan. The primary question of this work was:

> How does a more realistic treatment of forest patch size and shape in a fragmented and human-dominated landscape, through microclimate edge effects, alter calculations of the forest carbon balance using the ecosystem process model BIOME-BGC?

In order to address this question it was necessary to also explore:

> What are the spatial and temporal differences in air temperature in forest patch edges and interior in a particular human-dominated landscape, and how far into a typical forest patch do these microclimatic differences penetrate?

Completion of Chapters Two and Three identified a lack of high-resolution land-cover data at the parcel level. To better evaluate the effects of land-use development policies on land-cover change and ecosystem function, effectively combining the products of Chapters Two and Three, an analysis of land cover at the residential parcel level was necessary. To structure the collection of these data, Chapter Four answered the following research questions:

> What is the distribution of quantity and pattern of land-cover types in response to residential parcel size, in the exurban landscape of Southeastern Michigan? As well as, what is the degree of autocorrelation of land-cover quantity among residential parcels?

The aforementioned research questions and their corresponding chapters are contextualized within the field of land-change science and take a spatial approach and focus toward integrating data and process models. While I solely completed the research presented in this dissertation, unless otherwise cited, it was part of a broader project named SLUCE (Spatial Land Use and land cover Change and Ecological effects), funded by the National Science Foundation's Program on the Dynamics of Coupled Human and Natural Systems and conducted at the University of Michigan. Subsequent sections of this introductory chapter provide a brief description of the chapters that follow and their results. The conclusion chapter provides some insight into the future direction of this research and how it may come together into one unified framework for land-use policy and land-use and land-cover change analysis in exurban growth areas.

2. Evaluating the effects of privately versus publicly targeted land-use development policies on forest cover in Southeastern Michigan.

Traditional LUCC analyses have used remote sensing and GIS to describe land cover from local to global scales (Lillesand and Kiefer 1994). These approaches have generally been descriptive and have used statistical analyses to predict future LUCC trends. Unsatisfied with traditional static-type-of approaches, a number of researchers began using systems dynamics (Verburg et al. 1999), cellular automata (Deadman and Brown 1993, Clarke et al. 1997), and agent-based modeling to formally represent processes and behaviors that drive LUCC. Effectively, these contemporary efforts try to gain understanding and insight into what is more broadly referred to as coupled human-environment, coupled human-ecological, or coupled natural-human systems (Liu et al. 2007). From these contemporary approaches, agent-based modeling (ABM) has shown to "have a lot of potential to unravel some of the structural complexity of the system" (i.e. the land-use system). This is perhaps due to the ability of ABM to capture and integrate empirical data and traditional modelling methods with mechanistic or process based behaviours, feedbacks, and heterogeneity that are not analytically tractable using equation-based techniques (Parunak et al. 1998).

Typical agent-based LUCC models use existing agent-based platforms (e.g. Repast, Collier 2000; NetLogo, Wiliensky 1999). However, these platforms provide inadequate representation of spatial data and spatial functions. Conversely, spatial data modelling tools such as geographical information systems (GIS) or remote sensing software poorly represent time and process. As a result some researchers have built ABMs around existing GIS software (Deadman and Gimblett 1994, Gimblett 2002, Westervelt and Hopkins 1999). In an effort to improve ABM and GIS integration beyond theoretical linkages and loose couplings (Brown et al. 2005), Chapter Two presents an integrated GIS and ABM named DEED (Dynamic Ecological Exurban Development) that was created within a commercially available geographical information system (Robinson and Brown In Press).

The purpose of the DEED model was to evaluate privately versus publicly targeted land-use policies and their influence on forest cover in Scio Township, Southeastern Michigan (Chapter 2). Since only 5% of Michigan is publicly owned, it was hypothesized that privately directed land-use policies may be more effective than publicly directed ones as a method to increase forest conservation and subsequent carbon storage. A previously published residential location model named SOME (Brown and Robinson 2006) was incorporated into an entirely new conceptual framework. The new framework included township agents that set minimum lot-size zoning policies and developer agents, constrained by township policies, that evaluate the biophysical and geographic characteristics of farms and subdivide them into residential land-use parcels. The residential agents from the SOME model then evaluate the residential lots that become available by subdivision and settle at their most preferred location.

The model was used to evaluate the effects of two land-use development policies in isolation and in combination with one another. The first policy involved altering the lot-size zoning. Acting as an exclusionary policy, small lots and high density subdivisions were excluded from the model, followed by a subsequent scenario that excluded all but large lots and low density subdivisions. The second policy, a land acquisition policy, permitted the township agent to acquire 5% or 10% of the landscape for forest conservation. The location of these acquisitions also varied between three strategies: on farms with the least amount of forest in the township, on a set of randomly selected farms, and on farms with the highest levels of forest. Results from these scenarios 1) computationally verified literature demonstrating that larger lots lead to increased residential development or what may be called sprawling development patterns (e.g. Esparza and Carruthers) and that large lot-size zoning policies can influence the amount of forest cover (e.g. Munroe et al. 2005), 2) showed that when larger proportions of a township are placed in forest conservation there was an increase in the amount of forest cover in the township, regardless of lot-size zoning or conservation location strategies, and 3) when minimum lot-size zoning was applied in combination with land-acquisition, the rate of return on forest cover for areas placed in forest conservation was dependent on the location strategy used to locate those conservation areas (Robinson and Brown In Press).

3. Estimating carbon storage in fragmented and human-dominated landscapes.

As mentioned earlier, remote sensing is often used to quantify land cover at a specific date. In some cases it has been used to estimate ecosystem functions such as net primary production (NPP) and gross primary productivity (Zhao et al. 2007). However, few dynamic LUCC models have evaluated the ecological effects of LUCC or the ecological effects of the land-use policies they were designed to evaluate. One way to improve estimates of ecosystem function following LUCC involves the integration of LUCC analyses and models with well established landscape or big-leaf ecosystem process models (e.g. BIOME-BGC). However, these models do not incorporate patch edge or patch adjacency effects and when applied to two landscapes with the same amount of forest cover, but different land-cover patterns, they will produce identical amounts of carbon storage.

To address issues of patch edge associated with the size and shape of forest patches, fieldwork was conducted to measure the change in air temperature from the forest edge to the interior. Within-patch and landscape level air temperature measurements could then be used to reparameterize a model like BIOME-BGC to account for variation in patch shape and size. To collect these air temperature measurements three transects were established along the east-, south-, and west-facing aspects of an eastern deciduous forest patch in Southeastern Michigan (Chapter 3). Measurements were taken 15.25 m above ground at 15 m horizontal increments from the patch edge to a depth of 60 m from May 15th to August 31st 2006. A non-parametric Friedman repeated measures analysis of variance test revealed that estimated median air temperatures were significantly different ($\alpha = 0.01$) among transects (c.v. = 13.28) and for all measurement locations (c.v. = 30.58).

Field data and local National Weather Station data values were used to construct four air temperature treatments. The National Weather Station (NWS) data provided the first air temperature treatment and acted as a reference for comparison. Pearson's correlation coefficient statistics between the 2006 air temperature values measured interior and exterior to the forest with the NWS data were significant at $\alpha = 0.05$, 0.881 and 0.859 respectively. Given the high correlation, the difference between field-based and NWS data from 2006 was applied to the same year-day for historical data from 1930 – 2006. Three field-based air temperature treatments were constructed from the historical data and the air temperature data collected 1) exterior to the forest patch, 2) interior to the forest patch, and 3) from all field-based measurements that collectively define swaths or zones every 15 m from the forest edge to the interior. Respectively, these are referred to as the exterior, interior, and heterogeneous air temperature treatments.

Each air temperature treatment was applied to two computational experiments. The first experiment quantified forest carbon storage in the heterogeneous landscape of Dundee Township, Southeastern Michigan. Results from this experiment demonstrated that both within-patch and landscape heterogeneity in temperature values caused substantial differences in carbon storage estimates. Specifically, the interior, heterogeneous, and exterior treatments produced 318 080, 326 272, and 355 427 Mg C in Dundee Township at the end of the 77 year growth period (Robinson et al. Accepted). Therefore, within-patch air temperature heterogeneity produced an 8000 Mg C difference from the interior treatment and nearly a 30 000 Mg C difference from the exterior treatment. Collectively these within-patch measurements create a range of uncertainty of ~38 000 Mg C. If we include the reference treatment, which adds the effect of landscape heterogeneity in air temperature to the carbon estimates, the range extends to ~93 000 Mg C among all treatments.

To generalize results from the Dundee Township application, a second computational experiment was performed to evaluate the effects of landscape fragmentation on carbon storage using 12 hypothetical landscapes. Each homogeneous landscape was composed of 50% forest grown over 77 years in different fragmentation patterns based on the landscape metric known as the edge-to-area ratio. Since the area remained constant the homogeneous air temperature treatments (i.e. reference, exterior, and interior) produced different carbon storage estimates from each other, but because the area in forest did not change, fragmentation did not affect carbon estimates within each of these treatments.

Application of the heterogeneous air temperature treatment to increasingly fragmented landscapes provided more interesting results. As the landscape was altered from a single forest patch to complete fragmentation, carbon storage increased logarithmically to approximate the function y = 11898ln(x) + 586830 (R^2 = 0.92). Without fragmentation in the landscape, heterogeneous carbon storage values (12 393 Mg C) aligned closely with the interior air temperature treatment (12 350 Mg C). When the landscape was completely fragmented total carbon storage, under the heterogeneous treatment, had grown to 13 639 Mg C, which more closely aligned with total estimated carbon from exterior treatment (13 986 Mg C).

While the presented results verify existing empirical research that show increased forest productivity with increased temperature (Lieth 1975, Schlesinger 1997), the analysis and modeling of Chapter Three did not take into account a number of environmental factors or ecological processes that affect productivity. For example, within-canopy light penetration and attenuation or windthrow (Smithwick et al. 2003), tree mortality and the extraction of wood from a forest patch (Malanson and Kupfer 1993), erosion of soil organic carbon (Yadav and Malanson 2008), or disturbance regimes (Smithwick et al. 2007). Despite these omissions, the contents of Chapter Three illustrate the influence of human systems on forest growth and carbon storage. By quantifying these effects we can improve our understanding of the 1) impacts of human systems on ecological systems, 2) thresholds that when crossed may cause ecosystem failure, and 3) resilience and robustness of ecological systems to perturbations.

4. Residential land-cover characteristics in Southeastern Michigan.

Research conducted by members of the SLUCE project have advanced knowledge and understanding of the effects of integrating empirical research and agent-based models on spatial development patterns (e.g. Brown and Robinson 2006, Robinson et al. 2007, and Brown et al. 2008). Initially the group developed a simple residential location model, named SOME, that created residential household agents who used bounded rationality to select a settlement location based on distance to service centers and the aesthetic quality of a location. They used different aggregations of survey data from Marans et al. (2003) to investigate the effects of heterogeneity in variation and categorization on residential settlement patterns. Similar efforts to use survey data to empirically inform agent-based models have also taken place (e.g. Schreinemachers and Berger 2006). Collectively the land-use science community has been altering the use of ABMs as toy models or ‘proofs of existence’ (Waldrop 1990) that incorporate little or no empirical data toward empirically informed representations of decision-making actors and process that drive LUCC dynamics (Janssen and Ostrom 2006, Robinson et al. 2007).

The movement to more empirically informed agent-based models includes the incorporation of data for parameterization, calibration, and validation of both structural (e.g. type of decision-making strategy used) and outcome measurements (e.g. quantity of land cover). Having used the DEED model to evaluate land-use policy scenarios on aggregate forest cover at the township level, forest cover measurements were necessarily aggregated to the subdivision level because no data or literature could be identified that described the distribution of land-cover within household parcels. Furthermore, to link the DEED model with BIOME-BGC such that spatial within-parcel land-cover quantities and pattern affect carbon storage, requires knowledge of the distribution of land-cover types and patterns at the parcel level. In an effort to remedy this problem, the fourth chapter of this dissertation describes a simple land-cover analysis at the parcel level in Southeastern Michigan.

Results from the chapter provide an empirical description of the quantity, fragmentation, and autocorrelation of land-cover within residential land-use parcels. Specifically, it was found that the proportion of a property classified as forest increased with larger parcel sizes. The pattern of forest cover had more patches, increased patch sizes (on average), and increased forest edge, which collectively created less confined or compact forest patches with increasing parcel size. Furthermore, large parcels with high amounts of forest were more likely located adjacent to other parcels with similar amounts of forest cover. Results were different for each of the five land-cover types (i.e. forest cover, impervious, maintained lawn, other natural areas, and crop). For example, the shape and pattern of impervious surface remained relatively constant across different parcel sizes, while its proportion of the parcel area decreased with increasing parcel sizes. Maintained lawn was observed to have the most compact pattern of all land-cover types in residential parcels and its proportion of overall parcel area also decreased substantially with increasing parcel sizes.

By capturing the distribution and patterns of land-cover types across different parcel-sizes we can begin to understand the linkages between household land-cover behaviors, neighborhood interactions, and landscape patterns. While these data were not used to empirically inform an ABM as was done in Brown and Robinson (2006), the need for these data arose from the modeling efforts described in Chapters Two and Three and the data can be used to subsequently extend these modeling efforts.

5. Conclusions

The methods and results presented in this paper have provided steps forward for land-use and land-cover change research, which continues to gain importance in an arena of scientists trying to understand and mitigate global climate change. The approaches described can be applied to other regions and I have tried to foster the ability to transfer the DEED model to other sites by tightly coupling it to a commercial GIS, which is capable of swapping landscape data from other study areas.

Together these developments, by the author and the SLUCE project more generally, illustrate a step-wise progression of increasing complexity in the modelling of interacting processes and heterogeneous actors that influence land-use and land-cover change. The modelling process has been highly iterative, cycling between model development, data collection, and model refinement that will culminate in the creation of a GIS-based agent-based LUCC model that incorporates air temperature differences within forest patches to evaluate the effects of land-use policies on carbon storage. These advances in coupling empirical data (spatially referenced or not) with the process-oriented and mechanistic approach of ABM are necessary to improve land-use policy evaluation. Policy evaluations can be improved by this integrative approach because 1) the approach can integrate site specific data and behavior of actors who interact with the policy, and 2) the decision makers who create, change, or influence planning and land-use policies desire realism of model structure and predictions (Hayenga et al. 1968).

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Chapter 2 EVALUATING THE EFFECTS OF LAND-USE DEVELOPMENT POLICIES ON EXURBAN FOREST COVER: AN INTEGRATED AGENT-BASED GIS APPROACH

1. INTRODUCTION

Land acquisition is probably the most widely implemented ecologically-based land-use policy. Public land acquisition, by a governing agency (e.g. municipality), is used to conserve, preserve, or regenerate an ecosystem(s) and constrain the sale and development of that land. While costly in funds and management resources, conservation areas provide environmental and social benefits to both local and regional residents (e.g. Chiesura 2004). However, little land is actually set aside for public lands in most growing suburban and exurban landscapes. For example, publicly owned lands (which include campgrounds, dedicated open space, golf courses, preserves, parks, and federal lands) in seven counties of Southeastern Michigan (St. Clair, Livingston, Oakland, Macomb, Washtenaw, Wayne, and Monroe) accounted for only 64,739.4 ha or 5.4 % of the total 1,207,214 ha area (SEMCOG 2005). The low ratio of public to privately owned land suggests that land-use policies that influence private land use may provide a more effective forest-conservation strategy than public-land acquisition.

As a method of influencing private lands and their owners, land-use zoning policies were upheld by the Supreme Court of the United States in 1926 (Euclid 1926) to extend nuisance laws and have been used to plan for future development to separate land uses and reduce the number of land-use-based nuisances, substitute government-led collective property rights for individual property rights, and to protect and maintain low-density neighbourhoods (Nelson 1989). While debates on the efficiency or effectiveness of zoning have been long lived (Fischel 1980), minimum lot-size or exclusionary zoning has been shown to influence the amount of forest cover and placement of that cover within land parcels (Munroe *et al.* 2005). Furthermore, when minimum lot-size zoning policies are instantiated in combination with other land-use development policies they may have unintended consequences, specifically with regard to forest cover.

The evaluation of these types of land-use policies is difficult and requires consideration of both human and environmental complexity and associated uncertainties. The characteristics of such a complex system, i.e. heterogeneity, interaction, feedback, nonlinearities, and adaptation, all serve to complicate policy evaluations. The goal of this paper is to describe and illustrate the use of a GIS-based agent-based model (ABM), called the Dynamic Ecological Exurban Development (DEED) model. I use the model to generate a number of realisations of land-use and land-cover change that are the product of subdivision development and residential-location processes. Then, I evaluate how these processes are altered by minimum lot size zoning and land preservation strategies, both independently and in combination.

Recognizing that there are differences in the ecological quality and function provided by privately owned forested areas, secondary succession forest, and conservation areas set aside to protect existing forest or to establish new forest, I used forest cover as the primary measurement to evaluate land-use policies because it has both ecological and social benefits. Ecologically, increased forest cover reduces soil erosion (Dunn et al. 1993), reduces surface albedo (Pielke Sr. et al. 2002) , regulates local temperature through sensible and latent heat flux and evapotranspiration (Betts et al. 1996), increases available habitat, and increases carbon storage that, when aggregated over larger areas, can act to mitigate the effects of global warming. Similarly a number of social benefits derived from increased tree cover can be accrued through an increase in 1) the aesthetic quality of the landscape (Parsons and Daniel 2002), 2) privacy, 3) the filtering of air pollution and improved public health (Brack 2002), and 4) the quality of life for those living in proximity to forests.

In the next section I describe the study area to provide context for understanding how model components are informed by real-world actors and data. Then I introduce DEED and illustrate its use by analyzing different land-use development policies. Results of these policy experiments are then presented, followed by a discussion of the modelling process and the implications of these policy evaluations. Lastly I draw conclusions about the use of DEED for policy analysis.

2. STUDY AREA

The study area is the 88.8 km^2 area of Scio Township bounded by 42º15'11.9"N and -83º54'1.8"W, in the southwest, and 42º20'34"N and -83º46'44.6"W, in the northeast. Located in Michigan, USA, the township is encroached on by the city of Ann Arbor on its eastern edge and the Village of Dexter in the northwest. The township population increased by 29% between 1990 and 2000 (US Census 1990, 2001), making it one of only a few areas of Southeastern Michigan experiencing a high rate of growth. Similarly, the City of Ann Arbor experienced a 3.9% growth in population from 1990 to 2000 that made it the only city with positive growth in Southeastern Michigan. The proximity of the township to both urban and rural amenities and within-township access to major transportation routes (e.g. Interstate 94) provide a number of drivers recognised to increase population growth and urban sprawl.

Scio Township has a mixture of land covers and uses that have changed over time (Figure 2–1). Forest, other natural areas, and impervious surfaces have increased at the expense of crop land. Similarly, a decrease in agriculture is marked by an increase in residential land use and abandoned or undeveloped lands. The loss of agricultural land to low density housing, and urban sprawl is common in the Upper Midwest (Brown 2003). Since the township is highly fragmented and lies at Ann Arbor's urban-rural fringe, planning commissions are challenged to accommodate the conflicting needs of multiple actors who exploit public environmental resources.

While land-use changes in the township are the product of forces occurring inside and outside the township, planning in the township can profitably be viewed as a 'closed system' with little cooperation and coordination with other townships. The closed-system perspective is supported by legislation enacted at the state level (Gerber *et al.* 2005), like the Township Zoning Acts (PA 184 of 1943 and PA 168 of 1959), which foster a strong 'home-rule' policy, whereby the majority of power for land-use planning and policy resides at the township level (Wyckoff 2003). For the purposes of this paper, I bound the model to the limits of Scio Township, but incorporate influences from outside the township on the functioning of the model. I did not consider the possibilities of coordination and cooperation with neighbouring municipalities.

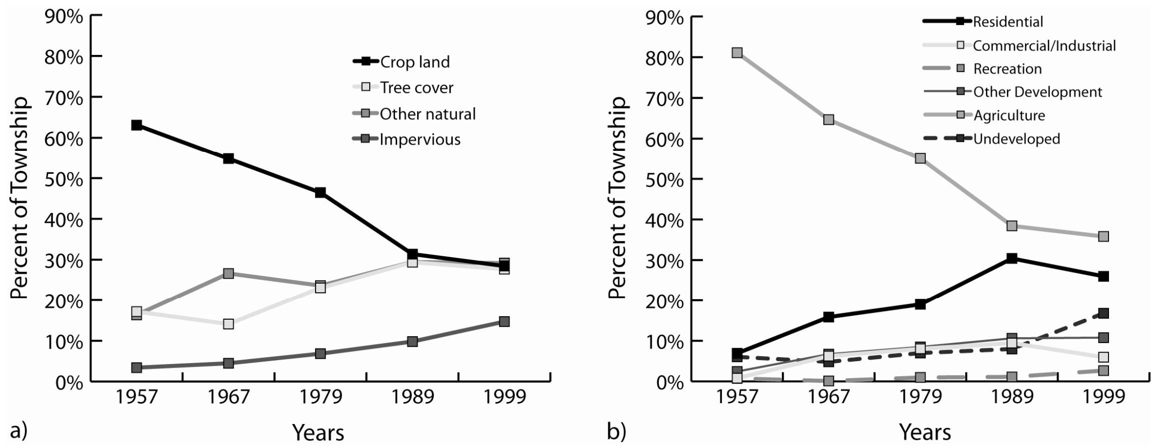

Figure 2–1: Land-cover (a) and land-use (b) trends in Scio Township from 1957-2000. Values calculated using air photo interpretation.

3. The Dynamic Ecological Exurban Development (DEED) Model

I use an ABM approach to simulate the decision-making behaviours, actions, and interactions of virtual agents that represent real-world actors or decision-making bodies and aggregate to produce landscape-level system outcomes. For a general overview and description of the approach see Parker et al. (2003) and Sengupta and Sieber (2007). I have co-authored previously published versions of the DEED conceptual model (Brown et al. 2008, Zellner et al. In Press) but altered it substantially by systematically integrating empirical research on agent behaviours and creating a new GIS-based ABM implementation for land-use policy analysis (Figure 2–2). In previous versions residential and developer agents used heuristic decision-making strategies that were based on expert opinion. In the presented version I integrate a residential location model based on survey data (Brown and Robinson 2006) and the results of a survival analysis (An and Brown 2008) to empirically inform the residential and developer agent decision-making strategies. Precursor tests using survival analysis to inform ABMs demonstrated that additional variables of influence were not included in previous versions of DEED (Brown et al. 2008) that I do include in the version presented in this paper. Furthermore, in this version of DEED I use a USDA defined method for creating landscape aesthetic maps for agent farm and subdivision-lot evaluation.

A model run begins by converting a grid of cells, representing hypothetical farm parcels, into farm agent polygons. Each farm-agent polygon is approximately 56.25 ha (140 acres). Statistics summarising landscape data by each farm polygon are stored as farm agent properties. Farm centroids are then created and the distance from each centroid to the nearest county road, to Detroit, and the nearest edge of land delineated as a city centre is stored in the farm properties. Next, a selection of farm parcels is acquired and placed in forest conservation by the township agent, which also implements the user defined policy scenarios. Developer agents then evaluate the biophysical and geographical characteristics of farms for subdivision. The subdivision type created by a developer agent has ecological effects on the landscape by altering the forest cover and creating different parcel lot densities. Residential household agents are then created; each one evaluates a bounded set of lots and acquires the deed to the lot that maximises the household's utility. Lastly, the township agent monitors and reports the amount of land use and land cover change that occurred. These and other model components are shown in Figure 2–3. The remainder of this section describes the landscape within which agents act and interact and then the characteristics and behavioural mechanisms for each agent type shown in Figure 2–3.

3.1 THE AGENT WORLD (LANDSCAPE)

The landscape is represented by land cover and soils data acquired from the Michigan Center for Geographic Information[1] (MCGI) and the U.S. Natural Resources Conservation Service (NRCS; SSURGO 2006)[2], respectively. Both datasets were resampled to 15-m resolution, where nine cells approximated the average minimum lot size (~0.5 acres) observed in several Southeastern Michigan townships (An et al. in review). Vector soil data were reclassified to prime versus not-prime farmland based on soil types defined as part of the Soil Survey of Washtenaw County conducted by the USDA and NRCS (Engel 1977). All DEED grids were 656 x 660 cells and incorporated *NoData* values to represent the township border. The township was composed of a total of 412,692 cells.

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[1] Michigan Center for Geographic Information: Department of Information Technology. URL: http://www.mcgi.state.mi.us/mgdl/, last accessed 1/16/2007.
[2] Soil Survey Geographic (SSURGO) Database, Soil Data Mart, Michigan, Washtenaw County. URL: http://www.ncgc.nrcs.usda.gov/products/datasets/ssurgo/, last accessed 1/16/2007.

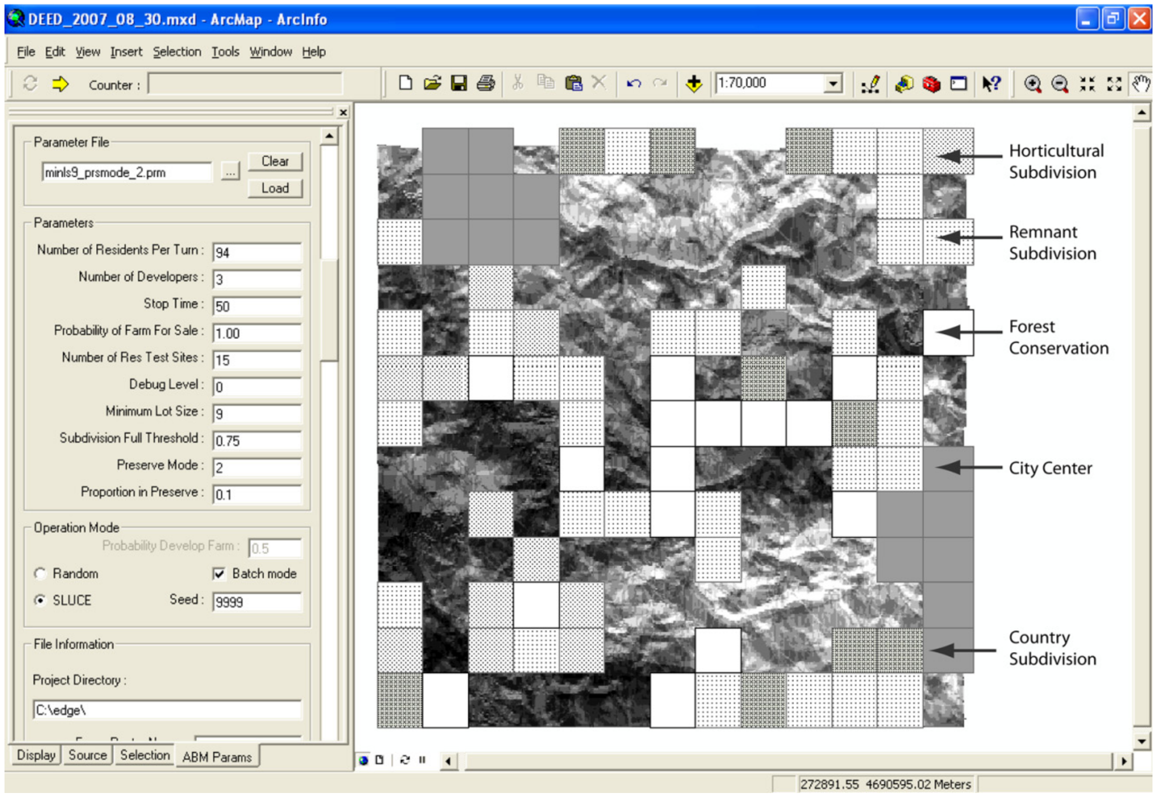

Figure 2–2: Screen capture illustrating a typical outcome from the unrestricted development scenario using the DEED model. A parameter window is integrated into the table of contents and a customized toolbar allows the user to start and reset the model.

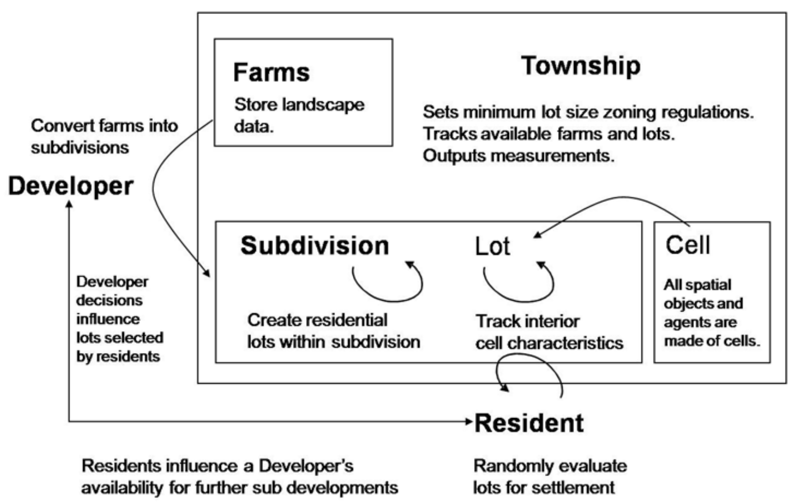

Figure 2–3: Outline of the interaction among modelled agents and objects as well as the primary behaviour of each. Farms, subdivisions, lots, and cells are all contained within the township agent.

Cell objects at each grid location store multiple values and allow for probing and altering landscape values at a high frequency without impeding model speed by frequent calls to raster grids. These objects also form the basis for land exchange among agents. For example, land holdings of farm agents are represented as a collection of cell objects that can be acquired by a developer agent who subdivides the farm into lots (i.e. smaller collections of cells). Deeds to lots are then transferred to residential household agents. Lastly the cell object framework allows for the extension of cell objects into automaton machines that have behaviours or landscape sub-models of their own (e.g. Box 2002, Deadman et al. 2004).

3.1.1 Landscape aesthetic quality

The measure of aesthetic quality across the landscape was created using a methodology developed by the USDA Forest Service (USDA 1971, 1995). The process involved first creating distributions of landscape character (e.g. lake areas and elevation values) and assigning quartiles of the distribution to one of three classes: minimal, common, or distinctive (Table 2-1). I also classified aspect, assuming greater sun exposure was preferred and north facing aspects did not contribute positively (i.e. value of 0), and included all rivers and public lands since few existed in the study area[3]. Next I calculated landscape visibility, as defined by USDA (1971, 1995), by measuring the Euclidean distance from cells to features of interest (i.e. distinctive lakes, rivers, and public lands) and classifying each cell into one of the following four visibility zones: immediate foreground, foreground, middle ground, or background area (USDA 1974, 1995, Table 2-1).

I then additively overlayed the landscape character and visibility grids and rescaled the resulting values to a range of 0-1 such that values close to one represented locations more aesthetically pleasing than values close to zero. Other methods may be used to derive an aesthetic quality map; however, I found no other useful literature beyond the USDA reports that provide a general evaluation of aesthetic quality that may be used over multiple locations. Furthermore, the incorporation of feature values into the

[3] The public lands dataset used was a subset of the conservation and recreation lands (CARL) dataset and was created by the Great Lakes Atlantic Regional Office, Ducks Unlimited, Inc. 1220 Eisenhower Place, Ann Arbor, MI 48108.

Table 2-1: Classification of landscape variability and visibility characteristics of Scio Township using the USDA landscape aesthetics and visual management system frameworks.

Aesthetic Quality Index Value	1	2	3	4
Landscape Character	Minimal	Common	Common	Distinctive
Elevation (m)	790 - 877	878 - 897	898 - 923	> 923
Lake Area (m^2)	260 - 1,142	1,143 - 2,098	2,099 - 5,755	> 5,755
Landscape Visibility (m)	> 6,437 = Background	805 - 6,437 = Middleground	92 - 804 = Foreground	0 - 91 = Immediate Foreground
Aspect (°)	Northeast (22.5 - 67.5) Northwest (292.6 - 337.5)	East (67.6 - 112.5) West (247.6 - 292.5)	Southeast (112.6 - 156.5) Southwest (202.6 - 247.5)	South (157.6 - 202.5)

landscape character classes is acceptable because I do not have enough precision in the data on residential preferences to acknowledge the more minor differences that occur in the continuous environmental data. The USDA methodology also incorporates user characteristics and preferences into their aesthetic quality map. Instead of directly including user characteristics and preferences into the map, I extracted them from a household survey and used the results from analysis of the survey to populate the characteristics and preferences of residential agents in the model (See section 3.2.3). In the next section I describe the agents that act within the landscape just described.

3.2 THE AGENTS

3.2.1 Township agent

The role of the township agent is to first implement a policy scenario based on user-specified parameters and then monitor and report on the holdings and transfers of farm, subdivision, and lot deeds at each time-step. The types of scenarios presented in this paper include 1) unrestricted development, 2) setting the minimum lot size zoning restriction on developers, 3) acquiring a number of farm properties for the creation of forest conservation areas, or 4) both 2 and 3 together. The township agent does not face a budget constraint on acquiring land for conservation. Instead the amount of land acquired is a parameter of the model, which I report values of municipal land acquisition at 0%, 5%, and 10% of the township area. When the township does acquire land for forest conservation, it pursues one of the following three strategies: locate conservation areas on 1) the least forested farms, 2) the most forested farms, or 3) randomly. Over time, the conservation area is managed to grow forest and reach a closed-canopy state by 20 years.

As the township agent acquires land for conservation purposes, it may force developers to substitute less preferred farms for subdivision. In addition to this type of substitution interaction between the township and developer agents, the township agent also determines if a developer can operate within the township via minimum lot-size constraints. Lastly, because the township maintains the lists of available farms and residential lots for acquisition it interacts with both developers and residential agents searching to acquire land.

3.2.2 Developer agents

Three developer agents are instantiated in the model, one to build each of three different kinds of subdivision developments. A developer agent begins by querying the township agent to determine if the residential density of the subdivision type it builds satisfies township building constraints (i.e. minimum lot size zoning). If not, then that agent is constrained from further action. If so, then the developer agent evaluates all non-developed farms based on its preferences for biophysical and geographical characteristics (Table 2-2) and subdivides the farm that maximizes its utility function. The developer agent may not partake in subsequent developments until the existing subdivision has been filled by residential demand, a threshold I set at 75%. The utility function used by developer agents to evaluate farms for subdivision takes the following form:

$$u_{d(farmx)} = \prod_{i=1}^{m} (\gamma_{i(farmx)})^{\alpha_{id}} + \sum_{j=1}^{n} (\gamma_{j(farmx)})\alpha_{jd} \qquad (1)$$

where $u_{d(farmx)}$ is the utility developer d receives from farm x; α_{id} is the preference weight developer d places on factor i; $\gamma_{i(farmx)}$ is the value of factor i at farm x; m is the number of non-binary factors; n is the number of binary factors evaluated. I separate the presence/absence or binary variables from non-binary variables because a single factor weight of zero would render a multiplicative utility function zero. Similarly, using exponents on values of 1 (presence of binary variable) would result in a 1, which would under-represent the significance of those variables. The above form overcomes both of these problems.

The preference weights for each developer agent are empirically informed using the relative difference among hazard rate coefficients, calculated by survival analysis (Klein and Moeschberger 1997). A hazard rate, representing the instantaneous risk that a parcel will be developed, is obtained by regression of the logarithm of the hazard against a linear combination of independent variables (those shown as landscape characteristics in Table 2-2, An and Brown 2008). I take the coefficients from the regression and rescale them (range 0 to 1) to represent the preference weight a developer agent has for each landscape characteristic. A single hazard rate equation is computed for each developer agent based on a sample of subdivisions and their interior lots, interpreted from aerial photographs taken at ~10-year intervals from 1950 to 2000 for 8 townships (Flushing, Oregon, Pittsfield, Putnam, Ray, Scio, Washington, and Woodstock) in Southeastern Michigan.

The subdivision typology was designed to represent three subdivision types found in Southeastern Michigan: country, horticultural, and remnant subdivisions (Nassauer personal communication, Brown et al. 2008). From the sample of subdivisions described above, the average lot size for each were 0.19 ha (0.48 acres), 0.82 ha (2.02 acres), and 1.26 ha (3.12 acres), respectively. Subdivisions also differ in their

Table 2-2: Hazard and preference weights (alpha values) for location characteristics influencing developer agent site selection decisions for subdivision. Bold numbers are significant at alpha = 0.10 level. Minimum lot sizes for country, horticultural, and remnant subdivisions are 0.19, 0.82, and 1.26, respectively.
Table adapted from Brown et al. (2008).

Location Characteristics	Unit	Country Subdivision		Horticultural Subdivision		Remnant Subdivision	
		% Δ in hazard rate per unit Δ	Scaled Alpha Values	% Δ in hazard rate per unit Δ	Scaled Alpha Values	% Δ in hazard rate per unit Δ	Scaled Alpha Values
Soil quality (prime farmland)	0 or 1	48.55	0.175	-3.05	0.103	**69.67**	0.511
Percent slope	1%	**60.65**	0.217	-12.81	0.000	**8.49**	0.082
Percent tree cover	1%	0.93	0.011	-2.06	0.113	**1.26**	0.031
Distance from county roads	1km	**-0.65**	0.005	-0.21	0.133	**0.13**	0.023
Distance from water	1km	**164.34**	0.576	24.24	0.390	46.69	0.350
Distance from nearest city	1km	-2.13	0.000	**4.74**	0.185	**-3.18**	0.000
Distance from Detroit	1km	2.64	0.016	**-5.60**	0.076	**-2.64**	0.004

respective amounts of forest cover. Specifically, the development of a country subdivision involves clearing all forest and little to no regrowth; horticultural subdivisions maintain pre-developed forest cover levels; remnant subdivisions may increase forest cover up to 4.05 ha (10 acres) after 10 years if less than 4.05 ha exists on the subdivision (An *et al.* in review). It is these differences in residential density and the biophysical outcomes of the subdivision process that drive much of the model results, which are shown in Figure 2–4.

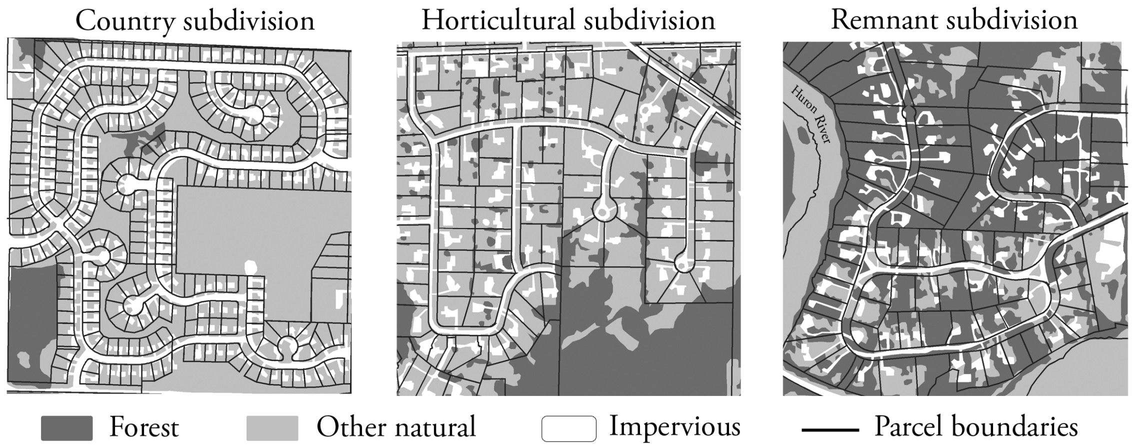

Figure 2–4: Digitized land-cover data and parcel boundaries for each of the three subdivision types used by the DEED model. Residential density decreases and forest cover increases as we move from the country subdivision (left) to the remnant subdivision (right).

The actual subdivision process as implemented in DEED is a rudimentary procedure that starts in the northwest corner of the farm and ends in the southeast corner. Each cell represents 225 m^2 (0.056 acres) and cells are allocated to lots sequentially until the size of the lot is equal to or greater than the observed average lot size for the subdivision type. The spatial pattern of lots within the subdivision is not an accurate representation of the pattern in the system of study. However, the analysis is at the subdivision level and therefore lot configurations do not influence the results.

3.2.3 Residential household agents

A factor analysis of a household location survey, administered in Southeastern Michigan, identified four factors influencing residential location: distance to schools/work, openness and naturalness, social comfort, and household characteristics (Fernandez et al. 2005). The first three factors operate at the scale represented by the model and were mapped into the following residential location drivers in a previously developed residential growth model named SOME: Euclidean distance to service centres (i.e. work, schools, and other urban amenities), aesthetic quality of a location, and neighbourhood similarity (Brown and Robinson 2006). The factor scores and their standard deviations are used to create distributions of values that are rescaled and used to populate residential household agent preference weights in DEED, as conducted by Brown and Robinson (2006).

At each time step of the model the township agent maintains a list of all available residential lots. Each residential household agent randomly chooses a number of those lots and from this subset selects the lot that maximizes the following utility function:

$$u_{r(x,y)} = \prod_{i=1}^{m} \left(1 - \left|\beta_i - \gamma_i\right|\right)^{\alpha_{(i,r)}} \qquad (2)$$

where $u_{r(x,y)}$ is the utility of the lot at location *(x,y)* for resident *r*; $\alpha_{(i,r)}$ is the preference weight the resident *r* places on factor *i*; β_i is the preferred value on factor *i* and assumed constant for all residents (i.e. all residents desire lower distance to service centres, higher aesthetic quality, and greater neighbourhood similarity); $\gamma_{i(x,y)}$ is the value of factor *i* from the lot at location *(x,y)*, and *m* is the number of factors evaluated. Measurement of neighbourhood similarity occurs by randomly selecting eight lots within the same subdivision as the lot being evaluated, based on the similarity of preferences between the searching household agent and those on eight randomly selected lots within the same subdivision those already settled[4]. Since the analysis is at the subdivision level the within-subdivision neighbour selection method does not affect the results.

The number of agents created by the model and populated by those survey data at each time step is based on GIS-derived residential building data. Residential building locations within the study area were digitised using aerial photo data from 1955, 1969, 1978, 1990, and 1998. A linear regression of the building data (excluding areas of Ann Arbor and Dexter) against year yielded the following equation: y = 291.2864 + 94.1773*(year), with an R^2=0.94, where y is the number of new households in a given year. Therefore, the model is calibrated to a fixed annual (time-step) increase of 94 residential structures (household agents) each year. Interaction among agents occur by (a) constraining future agent decision-making through the occupation of a lot and (b) influencing future agents' neighbourhood similarity measurements within the subdivision where a potential lot for settlement resides.

[4] Empty lots are assigned a neutral similarity value of 0.5, from the range of possible similarity values of 0.0-1.0. A value of 1 corresponds to perfect similarity and 0 corresponds to complete dissimilarity.

4. Computational experiments

I compared and contrasted the effectiveness of land-use policies designed to conserve forest cover by public-land acquisition versus lot-size zoning. To perform this comparison I held the number of new residents constant across all experiments and evaluated each policy independently, and then in combination, with the goal of identifying the most effective policy for forest conservation/restoration. Five scenarios are used to set the context for the policy experiments (i.e. unrestricted development, exclusionary zoning, land acquisition for conservation, conservation location strategy, and integration of lot-size zoning and land acquisition). Each experiment involves running the DEED model 30 times and averaging results to account for stochastic model behaviour.

I initialize the model with a hypothetical distribution of farm parcels that form a grid across the landscape. At each time-step of the model the developer agents evaluate the landscape characteristics of each farm that are initialized using the data described in Section 3.1. Ninety-four new household agents are created by the model at each time-step, based on the regression described in Section 3.2.3, and each household evaluates 15 random available lots for settlement before settling at the lot that maximizes the household agent's utility function. The process continues for 50 time-steps where each time-step represents one year, with a time-span chosen for a period that I have access to decadal land use and land cover data that will be used to validate higher fidelity and non-hypothetical versions of the model in the future.

The unrestricted development experiment indicates how the model performs in absence of lot-size zoning constraints or land acquisitions by the township. All three types of developers and their subsequent subdivisions exist (i.e. country, horticultural, and remnant) in this experiment, which result in a range of housing densities.

In the exclusionary zoning experiment, the township imposes two minimum lot-size zoning scenarios. The first scenario (EZ1) excludes the development of country subdivisions (i.e. high density developments) by setting the minimum lot-size zoning to parcels of 0.82 ha or greater. The second scenario (EZ2) excludes both country and horticultural subdivisions (i.e. high and medium density developments) by setting the minimum lot-size zoning constraints to parcels greater than or equal to 1.26 ha. In its essence, this experiment evaluates the trade-off between area of development and the type of development.

In the land acquisition for conservation experiment, zoning is removed and the township agent randomly acquires a number of farms for forest conservation. This experiment demonstrates how a fixed proportion of area in forest conservation may affect development and aggregate forest cover in the township. As noted earlier, public land ownership in the greater SEMCOG region, of which Scio Township is a part, approximated 5.4% in 2005. In extreme cases of public land acquisition, such as has been done to conserve panther habitat in Florida (Main et al. 1999), 10% of the land has been publicly acquired. Therefore, in this experiment I contrast the acquisition of 5% and 10% of the land area with the unrestricted and exclusionary zoning land use development policy experiments.

Extensive research exists on nature reserve selection and conservation location strategies (e.g. Prendergast et al. 1999). I extend the random preserve location strategy used in the land acquisition for conservation experiment with two simple forest conservation location strategies: 1) locate conservation areas on the least forested farms, and 2) locate them on the most forested farms.

To determine if there were significant interaction effects, I evaluated simultaneous changes in 1) minimum lot-size zoning, 2) proportion of the Township placed in conservation, and 3) strategy for locating conservation areas. I interpreted the results of the various policy combinations by calculating the amount of forest cover, the amount of development, and the *added-value* forest from each policy combination, where added value is the difference in forest cover between the unrestricted scenario and that obtained from a given scenario minus the fixed area placed in forest conservation.

5. Results

5.1 Unrestricted development

The lack of any policy resulted in the lowest amount of forested land and near lowest amount of developed land. Results from the unrestricted development experiment yielded an average aggregate forest amount of 15.2% of the township or 1332.87 ha in total (Table 2-3). Of the total township area, 17.2 % (1,523.63 ha) of the area, on average, was developed into some type of subdivision, of which 13 subdivisions were country subdivisions, 7 were horticultural subdivisions, and 6 were remnant subdivisions.

5.2 EXCLUSIONARY ZONING

Since the density of country subdivisions is higher than the other two subdivision types, one less country subdivision would mean 4 more horticultural subdivisions or 6 more remnant subdivisions, given a fixed residential demand. Due to these differences in subdivision densities, increasing the minimum lot-size to 0.82 ha (EZ1) tripled the area in residential land use to 52.1% from the unrestricted case of 17.16% (Table 2-3). Despite excluding the subdivision type that removes all forest from within its' boundary (i.e. country subdivisions), gains in aggregate forest cover under the EZ1 scenario were less substantial and led to only a slight increase from 15.2% to 16.7%, a 1.5 percent gain over the unrestricted case. When I excluded both country and horticultural subdivisions (EZ2 scenario) the area in residential land use exploded to 68.7% (6,102 ha) or four times the unrestricted case (17.16% or 1,524 ha). Similarly, only a modest gain in aggregate forest cover was achieved (EZ2, 17.5%) over the unrestricted case (15.2%). The added value of implementing these two land-use development policies on forest cover above the unrestricted case was 1.5% (133 ha, EZ1) and 2.3% (198 ha EZ2, Figure 2–5).

5.3 LAND ACQUISITION FOR CONSERVATION

The introduction of randomly located conservation areas appeared to have an unbiased substitution effect on developer behaviour. While conservation areas did occupy some lands sought by each developer type, using a random conservation allocation strategy had little effect on the amount of developed land (1626 ha) relative to the unrestricted case (1523 ha). In fact, randomly placing 5% and 10% of the township in conservation only increased developed lands on average by 103 and 111 ha (Table 2-3), respectively, which is less than the size of a single subdivision.

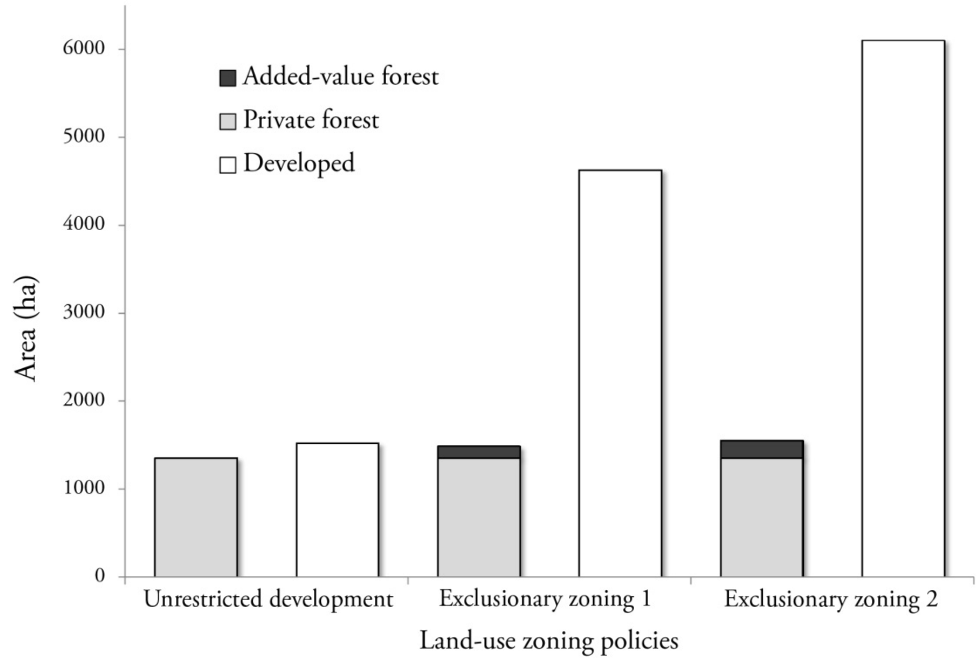

Figure 2–5: The resulting composition of forest and area developed under the unrestricted and exclusionary zoning land-use policy scenarios. Student t-test shows significant differences between all combinations of the three policy scenarios for both forest cover and residential development. Error bar values are too small to be visualized in this figure.

If we look at the changes in forest cover, we see that placing 5% of the township in forest conservation increased forest cover from the unrestricted case of 15.2% (1353 ha) to 19% (1689 ha). While this is a 25% increase in forest cover, perhaps it is more interesting to note that there was less than a 100% return on placing the area in forest conservation. We would expect that if the allocation of conservation areas was unbiased that we could have achieved the 15.2% in the unrestricted outcome + the 5% in conservation to produce 20.2% forest. Therefore while the township allocated a fixed amount of area (5%) in forest, developer behavior must have been altered to create less aggregate forest cover by the end of the simulation runs. This result is further evidenced when I placed 10% of the township in conservation and obtained only an 8.1% increase in forest cover over the unrestricted case (Figure 2–6).

Table 2-3: Results from all model runs and reported scenarios. CS = Country Subdivision, HS = Horticultural Subdivision, and RS = Remnant Subdivision. Values presented right of the minimum lot size column represent the average of 30 model runs using the same parameter setting. Area values are in hectares and values in parentheses are the average standard deviations for those runs. Cells with x represent an absence of that subdivision type under a given scenario.

Computational experiment	Conservation location strategy	Section	Min. lot size (ha)	Percent in conservation	Forest cover	Developed	Change in forest	Change in developed	Mean total area forested (sd)		Mean number of CS (sd)		Mean number of HS (sd)		Mean Number of RS (sd)	
Unrestricted		5.1	0.19	0%	15.20%	17.16%	0.00%	0.00%	1353.72	(1.65)	13.00	(0.00)	7.00	(0.00)	5.96	(0.18)
Exclusionary zoning	EZ1	5.2	0.82	0%	16.70%	52.15%	1.50%	34.99%	1486.50	(2.60)	x	x	35.93	(2.17)	44.26	(2.42)
	EZ2	5.2	1.26	0%	17.50%	68.72%	2.30%	51.56%	1552.12	(0.00)	x	x	x	x	106.00	(0.00)
Land acquisition for conservation	Random	5.3	0.19	5%	19.02%	18.31%	3.82%	1.16%	1689.46	(33.93)	12.80	(0.93)	7.40	(1.40)	7.50	(5.74)
		5.3	0.19	10%	23.30%	18.40%	8.10%	1.24%	2068.96	(34.91)	13.10	(1.11)	7.80	(1.68)	7.00	(4.98)
Conservation location strategy	Most forested	5.4	0.19	5%	16.87%	16.53%	1.67%	-0.63%	1497.81	(2.18)	13.83	(0.37)	5.16	(0.37)	5.96	(0.18)
		5.4	0.19	10%	19.66%	16.73%	4.46%	-0.43%	1745.79	(4.19)	13.96	(0.18)	5.40	(1.02)	5.83	(0.37)
	Least forested	5.4	0.19	5%	21.02%	30.00%	5.82%	12.84%	1865.97	(8.44)	10.00	(0.82)	11.53	(0.81)	24.40	(3.57)
		5.4	0.19	10%	25.98%	30.69%	10.78%	13.53%	2306.85	(5.01)	9.86	(0.34)	11.66	(0.47)	25.46	(1.61)
Combination Experiments	Most forested	5.5.1	0.82	5%	18.40%	51.51%	3.20%	34.35%	1633.97	(3.14)	x	x	36.03	(2.27)	43.07	(3.16)
		5.5.1	0.82	10%	21.28%	52.17%	6.08%	35.01%	1889.88	(2.81)	x	x	35.83	(2.08)	44.17	(2.73)
	Random	5.5.2	0.82	5%	20.43%	51.76%	5.23%	34.61%	1813.66	(29.09)	x	x	35.13	(2.01)	44.43	(2.45)
		5.5.2	0.82	10%	24.77%	51.54%	9.57%	34.38%	2199.17	(38.31)	x	x	35.30	(1.77)	43.97	(1.92)
	Least forested	5.5.3	0.82	5%	21.92%	51.87%	6.72%	34.71%	1946.25	(2.17)	x	x	33.93	(1.69)	46.27	(2.14)
		5.5.3	0.82	10%	26.87%	52.13%	11.67%	34.97%	2386.22	(2.09)	x	x	34.00	(1.73)	46.60	(1.98)
	Most forested	5.5.4	1.26	5%	19.25%	69.43%	4.05%	52.27%	1708.99	(0.00)	x	x	x	x	107.00	(0.00)
		5.5.4	1.26	10%	22.22%	69.43%	7.02%	52.27%	1972.66	(0.00)	x	x	x	x	107.00	(0.00)
	Random	5.5.5	1.26	5%	21.34%	69.17%	6.14%	52.01%	1894.61	(22.91)	x	x	x	x	106.53	(0.56)
		5.5.5	1.26	10%	25.66%	68.77%	10.46%	51.61%	2278.42	(29.12)	x	x	x	x	106.10	(0.65)
	Least forested	5.5.6	1.26	5%	22.50%	69.31%	7.30%	52.15%	1997.53	(0.00)	x	x	x	x	107.00	(0.00)
		5.5.6	1.26	10%	27.44%	69.31%	12.24%	52.15%	2436.87	(0.00)	x	x	x	x	107.00	(0.00)

Clues to why acquiring farms for conservation do not simply add a fixed amount for forest cover to the unrestricted scenario can be found in the high standard deviations associated with the number of country and remnant subdivisions (Table 2-3). By chance some simulations placed conservation areas on farms desired by the remnant subdivision developer. The remnant developer was then forced to substitute other farm locations that produced less desirable lots and because the quality of the environmental amenities were lower, residents moved to the more accessible country and horticultural subdivisions. Since country subdivisions clear forest cover and horticultural ones do not grow new forest cover, the random placement of areas in conservation do increase forest cover overall, but also decrease the amount obtained from larger lot subdivisions in the unrestricted case.

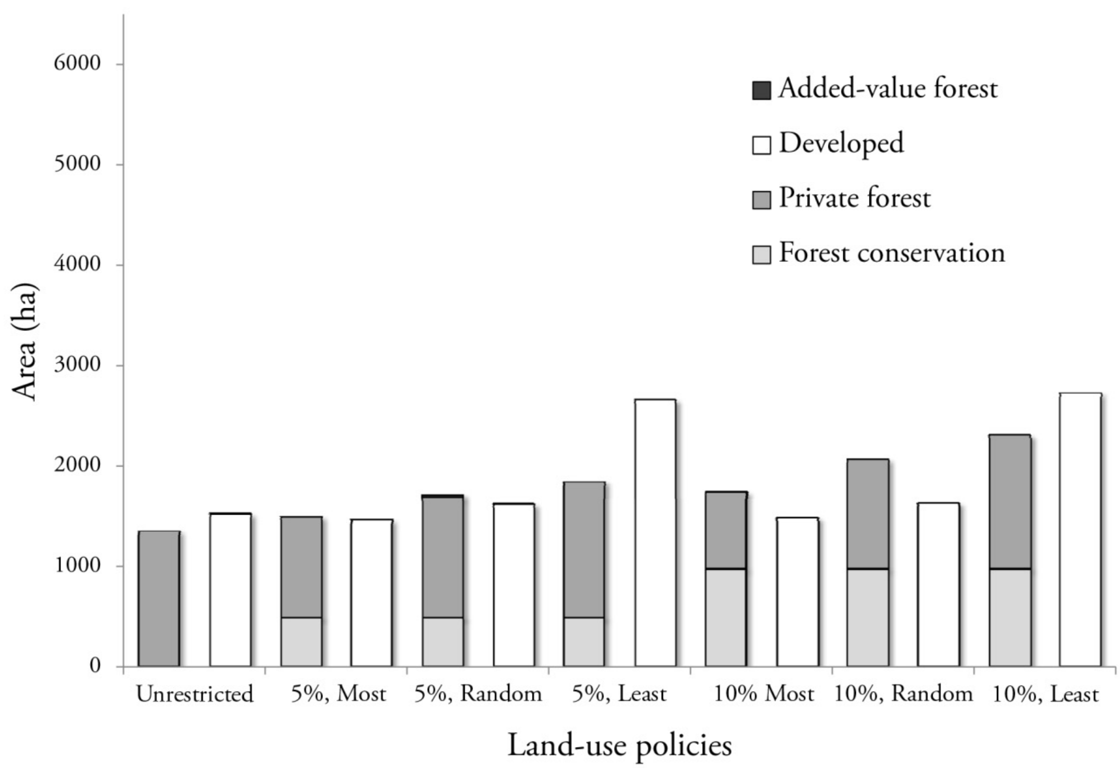

Figure 2–6: The resulting composition of forest and area developed under various land acquisition for forest conservation policy scenarios. Labels are read as Most = locating conservation areas on the most forested farms, Least = locating conservation areas on the least forested farms, Random = randomly located conservation areas, and 5% and 10% = proportion of the township acqured for forest conservation.

5.4 *Conservation location strategy*

Using a non-random strategy for locating conservation areas affected developer behaviour and, therefore, created different forest outcomes. Specifically, locating conservation areas on the most forested farms created the least effective strategy for maximising forest cover over the entire township (Table 2-3). The benefit of this strategy, however, was that it also led to the least amount of area in residential development. In contrast, locating the preserves on the least forested farms led to the highest forest cover outcomes and some of the highest levels of residential development.

When conservation areas were located on the most forested farms they tended to grow little new forest and forced remnant subdivision developers to substitute preferred farms for those less preferred. The substitution effect was so strong under this conservation location strategy, that the remnant developer experienced its lowest number of subdivision developments on average, relative to all reported scenarios, as well as the lowest level of variation in the number of subdivisions developed. Therefore, locating conservation areas on the most forested lands actually lead to the least amount of residential development or exurban sprawl within the scenarios reported (Table 2-3).

The overall forest cover results from this location strategy were 16.87% (1497.8 ha) and 19.66% (1745.8 ha) when 5% and 10% were placed in forest conservation, respectively. These scenario results hardly illustrate the utility of acquiring land for forest conservation on highly forested farms since they only produced a 1.6% and 4.4% increase in forest cover over the unrestricted case. Establishment of an additional 1.5% forest cover can similarly be achieved by increasing the minimum lot-size zoning to exclude country subdivisions (EZ1).

Results from scenarios that allocated conservation areas to the least forested farms are unique in that they created added-value forest, obtained high levels of forest cover within the township relative to other scenarios, and only created medium levels of residential development relative to other scenarios. Specifically, placing 5% and 10% of the landscape in conservation on the least forested farms produced aggregate forest levels of 21.02% (1866 ha) and 25.98% (2307 ha), respectively (Table 2-3**.** Both quantities of land acquisition resulted in more forest than the fixed amount set aside in conservation, i.e. an additional 567 ha and 1050 ha forested by choosing to locate conservation areas on the least forested farms. These results contrast those from allocating forest conservation areas on the most forested farms, which actually produced less forest cover than the fixed area in conservation plus the unrestricted scenario forest outcome.

Positive forest cover outcomes occurred because conservation areas located on farms with little to no forest were able to grow new closed-canopy forests within the time frame of the model. Also, generally the least forested farms are the most preferred locations for country subdivision development. Therefore, after the country subdivision developer agent achieves a couple of successful subdivisions, it is forced to substitute less preferred locations for those occupied by conservation areas. In doing so, the lots are less appealing and residents move towards the horticultural and remnant subdivisions, which maintain existing forest or may grow new forest. However, it is the success of some country subdivision developments that maintain medium to low levels of residential development or sprawl (approximately 30%) when either 5% or 10% of the township is placed in conservation, relative to other scenarios.

5.5 Integration of lot-size zoning and land acquisition policies

5.5.1 Exclusion of country subdivisions, acquisition of the most forested farms

Without country subdivisions the number of horticultural and remnant subdivisions jumped dramatically to ~36 and ~44 over the unrestricted scenario numbers of ~7 and ~6, respectively. Due to the larger minimum lot size and fixed residential demand, the area in residential development tripled that of the unrestricted development case to ~52%, but was not significantly different from the total developed area in the independent minimum-lot size zoning scenario EZ1. Placing greater area in forest conservation on the most forested farms did not affect the area developed but did affect the total aggregate forest cover in the township. However the effects were little with 5% and 10% in forest conservation leading to only 18.4% and 21.3% or a gain of 3.2% and 6.08% of the township forested over the unrestricted scenario. These results suggest that the combination of excluding high density country subdivisions and locating conservation areas on the most forested lands increased the amount of aggregate forest cover above either land-use policy implemented independently.

5.5.2 Exclusion of country subdivisions, random acquisition of farms

Randomly locating conservation areas in conjunction with a minimum lot-size zoning policy that excludes country subdivisions performed much the same as 5.5.1, but resulted in more aggregate forest cover. When 5% and 10% of the township was placed in forest conservation, the result was 20.43% (1814 ha) and 24.77% (2199 ha) forest cover or a 5.23% and 9.57% increase above the unrestricted scenario, respectively. The move to a random land acquisition strategy in conjunction with excluding country subdivisions improved upon the amount of aggregate forest cover obtained in the 5.5.1 combination policy as well as the independently applied EZ1, EZ2, and random and most forested farm conservation location policies.

5.5.3 Exclusion of country subdivisions, acquisition of the least forested farms

Surprisingly, changing the strategy for locating forest conservation areas again had little effect on the overall amount of residential development (~52%), much like the combinations in sections 5.5.1 and 5.5.2. Despite the small effect on developed area, the location strategy in combination with EZ1 resulted in 21.92% (1946 ha) and 26.87% (2386 ha) aggregate forest cover or a 6.7% and 11.6% increase in forest cover over the unrestricted scenario, respectively, when 5% and 10% of the township were placed in forest conservation (Table 2-3). Therefore, like the independent comparison of individual land-use acquisition strategies, locating forest conservation areas on the least forested farms in conjunction with excluding country subdivisions outperformed all other combination policies, as well as all isolated policies.

5.5.4 Exclusion of country and horticultural subdivisions, acquisition of the most forested farms

When I implemented a minimum lot-size zoning policy to exclude both country and horticultural subdivision developments in conjunction with acquiring land for forest conservation on the most forested farms we made only modest improvements on the level of forest cover with respect to the combination in section 5.5.1. Similarly, the outcomes with respect to forest cover 19.25% (1709 ha) and 22.22% (19.73 ha), when 5% and 10% of the township was placed in conservation respectively, were only improvements over the independent land acquisition policy (on most forested farms) and the EZ1 and EZ2 policies. All other policies produced greater amounts of forest cover and less area in residential development (Table 2-3).

5.5.5 Exclusion of country and horticultural subdivisions, random acquisition of farms

Like the previous combination of EZ2 and land acquisition (section 5.5.4), this strategy combination also had one of the highest levels of residential development at ~69% of the township area. Randomly allocating 5% and 10% of the township in forest conservation areas resulted in 21.34% (1895 ha) and 25.66% (2278 ha) of the township being forested. These results produced added-value forest since they created 6.14% and 10.46% more than the unrestricted case.

5.5.6 Exclusion of country and horticultural subdivisions, acquisition of the least forested farms

The last policy combination produced the largest amount of forest cover at both 5% and 10% in conservation, 22.5% (1998 ha) and 27.44% (2437 ha) respectively, and the second highest levels of residential development ~69%. The massive increased area in development ~18% over the combinations using EZ1 instead of EZ2 more than offset the minor gains achieved in forest area.

6. Discussion

6.1 Exurban forest cover scenarios

I began evaluating the effects of minimum lot-size zoning and land acquisition policies by establishing a baseline scenario void of either policy, which I called unrestricted development. In this scenario the model mechanisms were not constrained and produced hypothetical land-use and land-cover change results that were averaged over 30 simulations. Using the DEED model I compared 20 land use policies that each resulted in forest cover levels above that obtained by the unrestricted scenario. From these policies only one, acquiring land for conservation on the most forested locations, was able to increase aggregate forest cover and simultaneously reduce area in residential development.

As expected all policies that acquired land for forest conservation increased overall forest cover in the township. However, what was interesting was that 12 of the 18 policies that involved municipal land acquisition produced less than the expected amount of forest cover, where the expected amount was the unrestricted case plus the fixed area in forest conservation (Figure 2–7). The other six policies produced what I called added-value forest cover, yielding more forest cover than that obtained from the unrestricted scenario plus the fixed area in forest conservation. All scenarios that produced added-value forest were obtained by using a land acquisition strategy that located forest conservation areas on the least forested farms. I attributed the cause of these results to both a substitution effect and afforestation process. The substitution effect was the primary mechanism influencing model outcomes and occurred when a specific developer agent was forced to substitute less preferred farms for subdivision with those more preferred, but already acquired for conservation by the township. By acquiring land with little to no forest, the township was able to grow substantial forest cover while other forested areas had some probability of remaining depending on development patterns.

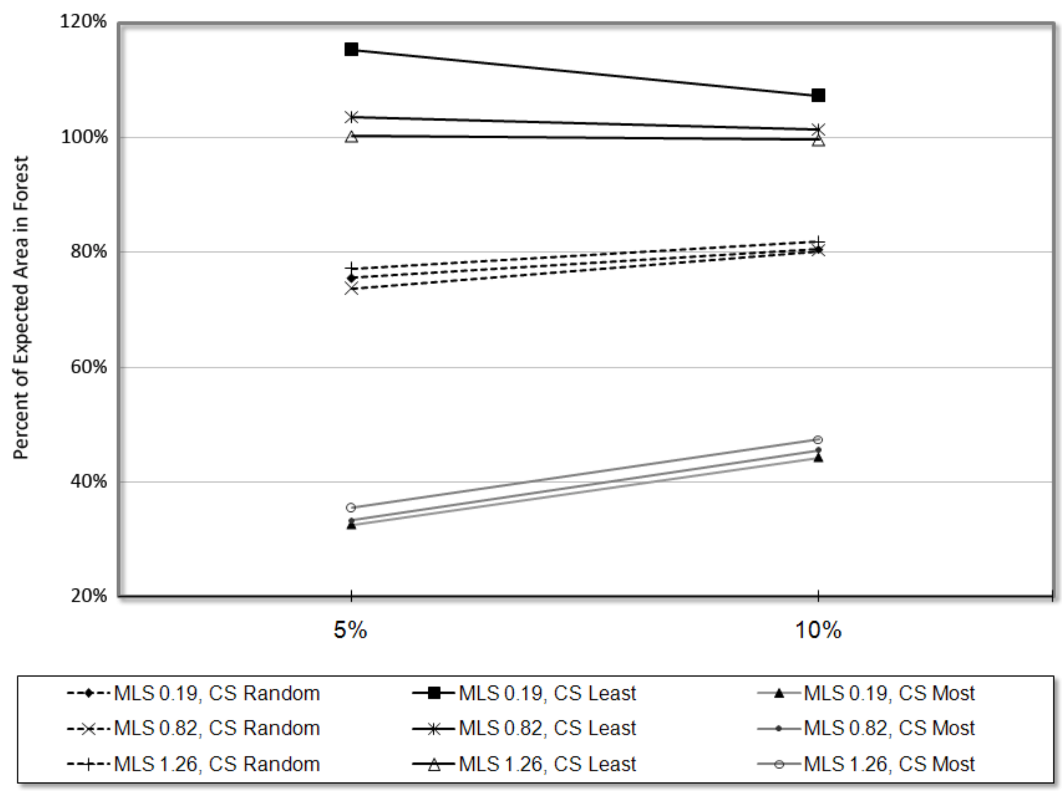

Figure 2–7: The percent of expected area in forest from different policy combination experiments. Acronyms are as follows: MLS = minimum lot size, CS = conservation location strategy, Least = least forested land, Most = most forested land and Rand = randomly located. For example, CS Most refers to the Conservation location Strategy, whereby conservation areas were located on the lands with the most forest.

Results from the scenario experiments computationally verified literature that show 1) large lot-size zoning policies lead to greater sprawl (e.g. Esparza and Carruthers 2000), and 2) large lot-size zoning policies can influence the amount of forest cover (e.g. Munroe et al. 2005, York et al. 2005), although I found this effect to be small relative to municipal land acquisition. I was surprised at the extent to which the location strategy affected the return on land acquisition for forest conservation. The location strategy for forest conservation land acquisition was more effective at increasing aggregate forest levels than the independent zoning policies (EZ1 and EZ2), the quantity of area acquired for forest conservation (5% and 10%), and any combination of the two (Figure 2–7).

6.2 Integrated approach over independent GIS approach

Results from using an integrated GIS and ABM framework for evaluating land-use development policies on forest cover provided insight into how those policies would act out over time and what aspects of those policies were more influential towards the goal of maximizing forest cover. The ability to play out scenarios or address 'what-if' questions through simulation is one of the key advantages of using agent-based modelling over traditional static accounts of change over time. For example, Taylor et al. (2007) evaluate the effectiveness of a local open-space ordinance at preserving natural features and rural character in subdivision developments using basic GIS mapping and spatial functions (e.g. buffering, overlay). While providing an excellent evaluation of an existing policy using historical data, their approach is unable to deal with feedback mechanisms and interaction effects that may influence the location, timing, and heterogeneous aspects of the residential developments affected by the policy. They note that their approach does not capture the dynamic transition between land-covers (e.g. open space to forest cover), which is due to the use of static temporal data (two time-steps in their analysis), and an inability to estimate how those processes will continue into the future.

It is these dynamic elements that I have attempted to bring to GIS by creating an integrated framework for evaluating land-use policy scenarios that may be used to better inform decision-makers before they make policy. Furthermore, it is possible to use an ABM approach to develop or generate potentially unknown policies, as was done by Zellner (2007) to generate policies for sustainable water use in Monroe County, Michigan. Lastly, the use of ABM in conjunction with GIS strengthens our ability to influence the actors making decisions on the landscape. We could potentially see how incentives (economic, social, or political) could alter land-user behaviours that subsequently alter land-cover quantity and placement.

6.3 Hypothetical world

The land-use and land-cover change modeling approach and analysis focused on how developer behavior, the subdivision process, and residential location are affected by exclusionary zoning and public land acquisition. I used empirical data to populate agent types and characteristics, to calibrate residential agent growth rates, and to provide initial land-cover and landscape aesthetic maps used in DEED. However, the initial conditions of the model (e.g. farm parcel configuration) are hypothetical and are not accurate to a specific time in reality. Furthermore, the missing processes, assumptions, and stochasticity built into the model inhibit the ability to validate aggregate model results. For these reasons my goal has not been a quantitative prediction of forest cover, but rather an assessment of the directional effects of different land use policies (i.e. DEED has been used as an exploratory model to investigate how different land-use development processes and policies act out over time and interact with each other). In its current state, the model can provide insight to land-use policy decision-makers, but it cannot provide predictive prescriptions for specific actions to take place.

To move the DEED model from an exploratory model to a predictive one would require additional mechanisms and real-world starting conditions. For example, although many of the farms were surveyed to160 acre parcels (Nelson 1995), the portion of a farm actually found in subdivision is much smaller. The model does not incorporate rural lots, which occupy large portions of land (on average 5 acres with a range up to ~15acres in the study region) and lead to scattered low density developments that may not be classified within a subdivision typology. Residential agents do not relocate, trade parcels, or directly compete for parcel acquisition. Similarly developer agents do not trade or compete for parcel acquisition. All land is assumed to be available for development and I did not incorporate dynamic on-farm processes such as cropping or abandonment, both of which may affect landscape perceptions.

Despite these constraints to validation, these initial experiments serve to partially verify the implementation of the conceptual model. For example, I anticipated that as smaller lots were excluded the amount of area in residential land use would increase, due to a constant supply of residential household agents at each time step and the lower residential density of the larger lot subdivisions. Similarly, the results would indicate a problem with the implementation if aggregate forest cover did not increase within the township as smaller lot sizes were excluded. This is because, in the conceptual model, forest cover is cleared for country subdivision development, maintained for horticultural subdivision development, and grown to a maximum of 4 ha of the subdivision area for remnant subdivision developments.

7. Conclusion

I illustrated the use of a GIS-based ABM, called DEED, in hypothetical scenarios using real-world data to produce results that describe the individual and interaction effects of minimum lot-size zoning and land-acquisition strategies on forest cover. The results of these scenarios illustrate some fundamental notions that may guide future research and application of land-use development policies. Specifically, I used an empirically-grounded model that incorporates the main actors influencing landscape change and the interactions between them and the environment to computationally verify existing research that suggests large lots lead to increased residential development or what may be called sprawling development patterns. I found that with larger proportions of a township placed in forest conservation there was a corresponding increase in the amount of forest cover in the township, regardless of lot size zoning or conservation location strategies.

When minimum lot-size zoning was applied in combination with land-acquisition, the rate of return on forest cover for areas placed in forest conservation was dependent on the location strategy used to locate those conservation areas. What I found was that when the location strategy placed conservation areas where there was little-to-no forest, conservation areas were able to grow additional forest. Furthermore, the occupation of these locations by the conservation areas decreased the appeal of country subdivisions that were forced to substitute less preferred locations for development. The appeal of the other subdivision types that maintained existing forest (i.e. horticultural subdivisions) or grew new forest (i.e. remnant subdivisions) increased in relative terms for many residential agents who moved into these subdivisions instead of the country subdivision type. Both of these mechanisms lead to increased forest growth.

The apparent synergies between the process modelling capabilities of ABMs and the data modelling and visualisation capabilities of GIS provide an opportunity for their integration and improved modelling-based research. The presented application demonstrates how a tight coupling of GIS and ABM can provide additional insight into how a dynamic land-use and land-cover system may be altered by different land-use development policies and how the process of evaluating those policies may be enhanced using this approach. Lastly, it is the authors hope that additional research integrating GIS and ABMs will facilitate ABM development by GIS users and help reduce barriers to entry to agent-based modelling.

8. ACKNOWLEDGEMENTS

Support for the presented research in the form of grants was provided by the National Science and Engineering Research Council of Canada (NSERC), the Graham Environmental Sustainability Institute (GESI) at the University of Michigan, and the National Science Foundation Biocomplexity in the Environment Program (BCS-0119804) for Project SLUCE. I would like to acknowledge with gratitude the intellectual support of my colleagues Li An, Rick Riolo, Moira Zellner, William Rand, Scott Page, Joan Nassauer, Bobbi Low, Qing Tian, Jason Taylor, and Michael Norton. Lastly, I would like to thank my co-author Dan Brown, three anonymous reviewers, and the editor for assisting in the publication of this chapter in the International Journal of Geographical Information Science.

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Chapter 3 MODELLING CARBON STORAGE IN HIGHLY FRAGMENTED AND HUMAN-DOMINATED LANDSCAPES: LINKING LAND-COVER PATTERNS AND ECOSYSTEM MODELS

1. INTRODUCTION

Processes of land-use and land-cover change (LUCC) are characterized by local complexities and feedbacks that produce global consequences (Foley et al., 2005), including effects on climate (Houghton et al., 1999; Schimel et al., 2000; Barford et al., 2001). The alteration of the earth's surface changes the albedo (Pielke Sr. et al., 2002); sensible and latent heat flux; evaporation (Betts et al. 1996); biodiversity (Poschlod et al., 2005); biophysical characteristics that contribute to nutrient and hydrological cycling (Hubacek and Vazquez, 2002); and carbon (C) storage (Dixon et al., 1994). Each of these biophysical functions significantly influence global climate (Riebsame et al., 1994). Globally, land-use change in the 1980s and 1990s contributed 1.4[5] and 1.6 Pg C yr^{-1} to the atmosphere (1 Pg = 10^{15} g = 1 Gt) and represented approximately 30% of anthropogenic efflux of carbon to the atmosphere (Dixon et al., 1994). Conversely, mid-to-high latitude forest expansion driven by reduced agricultural land use in the 1990s (Gower, 2003) contributed to a net carbon sink by land-use within these regions (Fan et al., 1998; Caspersen et al., 2000).

Because LUCC is complex and driven by human activities, understanding its' effects on ecosystem processes involves studying a coupled human-environment system. To date, a number of projects have determined the dominant mechanisms influencing these coupled systems and in some cases mapped (with measured error) observed spatial patterns of LUCC (e.g. Deadman et al., 2004; Huigen, 2004). However, most LUCC projects fall short by failing to incorporate measurements of key ecological functions (e.g. biogeochemical cycling) and how those functions affect and are affected by human systems. It is necessary to represent both of these types of interactions if we wish to

[5] Land-use change flux based on Chapter 7 of the 2007 Intergovernmental Panel on Climate Change (IPCC) report, which noted the land-use change induced carbon efflux to the atmosphere to be 1.4 (0.4 - 2.3) Pg C yr^{-1} for the 1980s and 1.6 (0.5 - 2.7) Pg C yr^{-1} for the 1990s. Values in parentheses represent range of uncertainty.

assess the influence of local LUCC on global climate change. This problem is recognized by a number of government agencies and affiliations (e.g. Turner et al., 1995; Lambin et al., 1999; Gimblett, 2002; Parker et al., 2002; Lobo, 2004; Gutman et al., 2004; Ojima and Moran et al., 2004).

Paralleling LUCC initiatives are ecological studies that use models to explore the effects of succession, disturbance, competition, biophysical changes, and geography on ecosystem structure, function, and biodiversity (Parton et al., 1987; Jeltsch et al., 1996; He et al., 1999a,b; Gustafson et al., 2000; Shugart, 2000; Urban and Keitt, 2001; Howe and Baker, 2003). Ecosystem process models that focus on biogeochemical cycling have found utility in global climate change research because they are typically applied at resolutions $\geq 1\ km^2$ and can quantify evapotranspiration, water use efficiency, carbon (C) and other nutrient pools and fluxes. In contrast to the coarse-scale resolution ($\geq 1\ km^2$) typically employed by ecosystem process models, some models such as BIOME-BGC or 3-PG have been successfully applied at a resolution of 30m (Coops and Waring, 2001). While successfully applied at relatively fine resolutions, few if any such models incorporate the effects of ecosystem patch edge, shape (e.g. irregular forest patch perimeters), size, or edge-to-area ratios on ecosystem function. Therefore, applying a model such as BIOME-BGC to simulate forest C in two landscapes with equal forest area, but different spatial pattern, would produce equal amounts of total forest carbon.

However, microclimate and biophysical characteristics are altered along a transition zone between the adjacent ecosystem (e.g. prairie) and the forest interior (Matlack, 1993) on the scale of tens to hundreds of meters. On exposed forest patch edges, light and wind may penetrate beneath closed canopies causing gradient changes in temperature, moisture, and the vapour pressure deficit deep into the forest (Chen et al., 1995). The depth of penetration and alteration of climate characteristics is a function of forest type, structural characteristics (e.g. stem density), aspect, and side-canopy presence (Matlack, 1993). For eastern deciduous forests these effects have been observed to penetrate into patches in the range 15-50 m, while edge-effect penetration has been observed to be greater than 240 m in Douglas fir forests (Chen et al., 1995).

In addition to edge effects, landscape heterogeneities in the form of patch shape, patch size, landform, and proximal land use and land cover may influence local climate.

For example, the perimeter/area ratio of a patch along with patch size can describe the degree of core area of a forest patch that is buffered by local climate (Collinge, 1996). The physical characteristics of the landscape (i.e. landform) such as slope and aspect influence the degree of incident solar radiation; elevation influences adiabatic processes; and proximity to water features can influence humidity levels, each of which can affect local climate (Rosenberg et al., 1983). Similarly, different types of land uses and land covers have also been shown to influence local climate (Landsberg, 1970). For example, urban heat islands have extended influence on temperature values beyond city limits (Arnfield, 2003) and agricultural lands can affect heat fluxes and influence thunderstorm frequency (Raddatz, 2007).

As a step towards integrating LUCC and ecosystem models focused on biogeochemical cycling, I addressed issues of edge and landscape heterogeneity, whereby the landscape is composed of forest patches of variable shapes and sizes within a matrix of other types of land cover. This research addresses two specific questions through analysis of ecological field data, its subsequent incorporation into BIOME-BGC, and application to the heterogeneous landscape of Southeastern Michigan. The primary question is: How does a more realistic treatment of forest patch size and shape in a fragmented and human-dominated landscape, through microclimate edge effects, alter calculations of the forest carbon balance using the ecosystem process model BIOME-BGC? In order to address this question I additionally explore: What are the spatial and temporal differences in air temperature in forest patch edges and interior in a particular human-dominated landscape, and how far into a typical forest patch do these microclimatic differences penetrate?

2. Materials and methods

2.1 Study area

The study area was located within the heterogeneous, fragmented, and human-dominated landscape of Dundee Township, Southeastern Michigan. Forest patches in this region primarily exist as secondary growth forest, remnants of abandoned agricultural land. Agriculture peaked as a land use in the area in the late 1880s to 1900 and has declined from 1910 to the present. Since then, heterogeneity of the landscape has been increasing due to LUCC at the urban-rural interface. While the region is representative of land-use histories and land-cover patterns of the Midwest and populated regions in the eastern temperate zone of North America, I chose a single township (i.e. Dundee Township, in Monroe County, Southeastern Michigan, USA) 12,577 ha in area, as the study area to conduct both field work and model application (Figure 3–1).

In 2001, approximately 10% (1277.28 ha) of the study area was forested (Homer et al., 2007 - NLCD data). The amount of forest in Dundee Township was below average for Southeastern Michigan where the mean during the same time period was 28% forest (standard deviation 13%) based on 140 townships sampled from the 10 adjoining counties (Genesee, Lapeer, Lenawee, Livingston, Macomb, Monroe, Oakland, St. Clair, Washtenaw, and Wayne). Dundee Township illustrates the extent to which the regional landscape has been modified, fragmented, and become dominated by anthropogenic land uses. The township is now composed of 262 forest patches in an agricultural and residential matrix with a mean forest patch size of 5.02 ha (standard deviation 10.45 ha). The total patch edge in Dundee Township is 398 km and the edge density (total edge / total landscape area) is 31.64 $m \cdot ha^{-1}$.

From within Dundee Township I chose a single forest patch, typical of the region, to conduct the field study. The forest patch provided ideal characteristics to study changes in daily minimum and maximum temperatures from the forest patch edge inward. The field study patch consisted of a single, privately owned, eastern deciduous forest patch that was situated (1) 0.5 km from the nearest creek and 0.7 km from other tree cover, (2) on a slope less than 3˚, and (3) within a uniform surrounding vegetation (i.e. corn and soy crops). The edges of the selected patch were linear, partially closed side-canopy, and perpendicular to the cardinal directions. The 220 m x 210 m (4.62 ha) forest patch was approximately 80 years in age with a canopy height of 24-30 m. The dimensions of the patch ensured that measurement points were not located where the influence of two adjacent edges could overlap. To the best of my knowledge the patch had not experienced any significant major disturbance in the past 80 years, although some wind throw is a normal part of the disturbance regime in this region (Frelich, 2002) and did occur along two of the three transects during the study period.

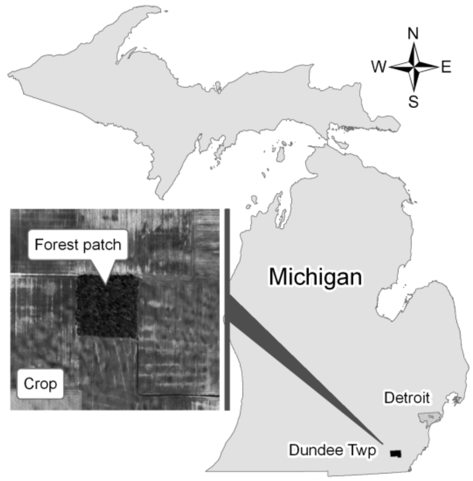

Figure 3–1: Aerial photograph of the study forest patch (year 2000) and location of the Dundee Township study area in Michigan, USA.

The field study patch was located within the Maumee Lake Plain ecosystem type (Albert, 1995), which runs across much of the eastern boarder of Southeast Michigan. Forests in this ecosystem are characterized by beech-sugar maple or elm-ash species (Albert, 1995). The loamy sand on which the site is located was produced from glacial outwash sand and gravel, postglacial alluvium, and coarse-textured till from end moraines. These soil and surficial geologic conditions are typical of Southeastern Michigan.

2.2 Field data collection and analysis

The objective of the field-data collection and analysis was to statistically determine if there was a difference in minimum and maximum air temperature among locations from the edge of an eastern deciduous forest patch inward to forest interior. I performed two analyses to test whether air temperature differed with forest depth. To accommodate repeated measures of more than two dependent samples, I first performed a non-parametric Friedman two-way analysis of variance by ranks test (Sheskin, 2004). Using observed air temperature as the response variable, sensor locations as the treatments, and observations blocked by day and hour, I calculated a chi-square (χ^2) value for maximum and minimum air-temperature observations for each transect and all sensors combined. Second, I performed a functional data analysis to further illustrate the direction of the temperature gradient from the forest patch edge inward to the forest interior. A functional data analysis was conducted by creating a simple linear model of temperature as a function of distance for each point in time, which took the following form:

$$T_{obs} = a + m(d - d_{ave}) \tag{1}$$

where T_{obs} is the observed temperature, a is the intercept, m the slope, d the distance the temperature measurement was taken from the forest edge into the interior, and d_{ave} is the average of the distance measurements. I performed a linear regression of the six temperature values (i.e. taken at 0, 15, 30, 45, 60 m, and at the forest interior 80 m) to model the change in slope and intercept with respect to distance for each hourly measurement.

Field-data collection focused on maximum and minimum air-temperature observations because these two climate parameters can be used by the MTCLIM model (Running et al., 1987; Thornton and Running, 1999), which accompanies BIOME-BGC, to produce the climate variables needed to execute BIOME-BGC (i.e. minimum and maximum temperature, daylight average temperature, short-wave solar radiation, vapour pressure deficit, and day length). Three transects corresponding to east, south, and west aspects were established at the forest patch edge extending to a depth of 60 m within the forest patch. Due to a number of limitations I did not evaluate the edge effects from the northern edge; however other studies have suggested that there is little to no effect on the northern edge in the northern hemisphere (Matlack, 1993). Transects were located beyond 60 m from a patch corner to prevent edge overlap. Existing literature suggests that abiotic edge gradients often cease to extend beyond 50-60 m in eastern deciduous forests (Matlack, 1993; Cadenasso et al., 1997). At each measurement point, temperature was recorded at 15.25 m (50 ft) aboveground and approximately 1 m from the stem on the south side of a live tree. Temperature measurements were recorded every 15 minutes at the forest edge and distances approximating 0, 15, 30, 45, and 60 m inside the forest along each transect. The data reported in this paper include hourly maximum and minimum air temperature for all hours from 15 May to 31 August 2006, using Hobo Pro Temp/RH data loggers produced by Onset (www.onsetcomp.com).

An additional measurement point was located within the forest at a depth of approximately 80 m from the south-facing edge (to provide a measure of patch interior temperature) and a second additional measurement point was placed external to the forest patch to replicate standard meteorological measures of air temperature. The external measurement point was located between two agricultural fields (soybean and corn) along a section of turf grass used for transportation by the property owner. The external sensor was located approximately 4.5 m above the ground on a wooden pole erected away from shade and above the average fully grown corn stalk of approximately 2.5 m. Each sensor was covered by a rain shield to prevent direct contact by sunlight and moisture.

In addition to air-temperature measurements, a number of independent site variables were recorded along each transect and surrounding each forest canopy measurement point. These variables included distance from edge, basal area of trees with a diameter at breast height (dbh) > 1.5 cm (conducted on 17 September 2006), and leaf area index (LAI). A number of 10 m x 10 m plots were established, centered on each forest canopy measurement point, in a line inward from the patch edge. Plots located on the patch edge had a dimension of 10 m x 5 m. Leaf area index (LAI) measurements were taken on 18 August 2006, using an LAI 2000 plant canopy analyzer produced by LI-COR Biosciences. Three readings external to the forest, using a 180˚ lens cap to block residual backscatter from the forest, were taken first. Then eight readings were taken within each plot from which an average LAI value was calculated (LI-COR, 1992). This process was repeated for each plot.

Individual tree biomass was estimated using allometric equations of the form $M = aD^b$ where M is the oven-dry weight (kg) of biomass, D is the dbh (cm), and *a* and *b* are parameters based on previous empirical research (Ter-Mikaelian and Korzukhin, 1997). I used two methods to extrapolate biomass values up to the patch level and then divided the result by the patch area to obtain biomass in $kg \cdot m^{-2}$ (Table 3-1). Content of carbon was calculated as 45% of the oven-dry biomass (Whittaker, 1975) to coincide with previously published values; however, others have shown percent carbon can be higher (e.g. Currie et al., 2003; Gower, 2003). GPP values for Southeastern Michigan were obtained at a coarse resolution of 1 km^2 using remote sensing techniques (Zhao et al., 2007). The authors used the light use efficiency (LUE or ε) from BIOME-BGC to estimate GPP and derived a value of 759 g $C \cdot m^{-2}$ in 1999 (Zhao et al., 2007).

Table 3-1: Above ground biomass and carbon (C) content measurements for eastern deciduous broadleaf forests.

Source	Location	Dominant Overstory Species	Forest Type	Biomass $kg \cdot m^{-2}$	Carbon $kg \cdot m^{-2}$	NPP $Mg\ C \cdot ha^{-1} \cdot yr^{-1}$	LAI $m^2 \cdot m^{-2}$
Botkin et al. (1993)[a]	Eastern North America	Across 13 physiographic regions	Temperate Deciduous	8.1 ± 1.4	3.6 ± 0.6	—	—
Botkin et al. (1993)[b]	Eastern North America	Across 13 physiographic regions	Temperate Deciduous	9.8 ± 1.8	4.4 ± 0.8	—	—
Botkin et al. (1993)[c]	Eastern North America	Across 13 physiographic regions	Temperate Deciduous	9.2 ± 1.6	4.1 ± 0.7	—	—
Bolstad et al. (2001)	Coweeta Hyrologic Laboratory, Western North Carolina, USA	Chesnut oak (Quercus prinus), Red Maple (*Acer rubrum* L.), (*Catya*), and Amerian tulip tree (*Liriodendron tulipifera*).	Eastern North American Deciduous	—	—	9.20	2.7-8.2
Curtis et al. (2002)	Walker Branch, Eastern Tennessee, USA	White oak (*Quercus alba* L.), Red maple, Sugar maple (*Acer saccharum* Marchall.), and American tulip tree.	Eastern North American Deciduous	21.63[e]	9.73	5.39	6.2
Curtis et al. (2002)	Morgan Monroe State Forest, South-central Indiana, USA	Sugar maple, Yellow poplar, Sassafras (*Sassafras albidum* Nutt.), White oak, and Black oak (*Quercus velutina* Lam.)	Eastern North American Deciduous	22.65[e]	10.19	5.29	4.9
Curtis et al. (2002)	Harvard Forest, North-central Massachusetts, USA	Red oak (*Quercus rubra* L.), Black oak, Red maple, Hemlock (*Tsuga canadensis* L.), White pine (*Pinus strobus* L.) and Red pine (*Pinus resinosa* Aiton.) plantations.	Eastern North American Deciduous	23.34[e]	10.50	3.20	4.0
Curtis et al. (2002)	University of Michigan Biological Station, Northern Lower Michigan	Bigtooth aspen (*Populus grandidentata* Michx.), Trembling aspen (*Populus tremuloides* Michx.), Red oak, Beech (*Fagus grandifolia* Ehrh.), sugar maple, white pine, and hemlock.	Eastern North American Deciduous	13.83[e]	6.22[d]	3.38	3.7
Curtis et al. (2002)	Willow creek, North-central Wisconsin, USA	Sugar maple, American basswood (*Tilia americana* L.), green ash (*Fraxinus pennsylvanica* Marsh.), and red oak.	Eastern North American Deciduous	17.47[e]	7.86	3.00	4.2
Newman et al. (2006)	Southeastern Kentucky, USA	Cucumber magnolia (*Magnolia acuminata*), American tulip tree, Sugar maple, American basswood, Red oak, Red maple,	Mesic Deciduous	24.08[e]	10.83	5.13-11.56[h]	7.9
Newman et al. (2006)	Southeastern Kentucky, USA	Chesnut oak (*Quercus prinus*), Scarlet oak (*Quercus coccinea*), Black oak (*Quercus veluntina*), Red maple, Red oak, White oak.	Xeric Deciduous	24.68[e]	11.10	5.13-11.56[h]	3.5
Average				**17.48**	**7.85**		**4.3**
This study[f]	Southeastern Lower Michigan, USA	American basswood, American elm (*Ulmus americana*), Red maple, Swamp white oak (*Quercus bcolor*), Hawthorn (*Crataegus monogyna*), Red oack, Bitternut hickory (*Carya cordiformis*), White ash (*Fraxinus americana*), and Silver maple (*Acer saccharinum*).	Eastern North American Deciduous	19.12	8.61	—	3.36
This study[g]	Southeastern Lower Michigan, USA		Eastern North American Deciduous	21.44	9.65	—	3.26

[a] Values of Botkin et al. (1993) are lower than others due their random sampling which included young forests.
[b] Use of general hardwoods biomass equations from Clark et al. (1986a) on all angiosperms. [c] Use of general hardwoods biomass equations from Clark et al. (1986b) on all angiosperms.
[d] Foliage and understory omitted. [e] Calculated from reported C values [f] Average of subplots. [g] Total forest patch average. [h] Range over several sites - No available data reported

2.3 CALIBRATION AND PARAMETERIZATION OF *BIOME-BGC*

My objective to integrate land-cover data and forest ecosystem processes at a fine resolution (15m) was operationalized by using BIOME-BGC. The model was developed to determine if a single ecosystem process model could be useful for representing biogeochemical cycling in multiple biome types (Running and Hunt, 1993). BIOME-BGC simulates multiple C storage and flux outputs and partitions storage and respiration into individual pools (e.g. canopy, stem, and roots) as well as ecosystem level outputs such as gross photosynthesis and net primary production (NPP) (Running and Coughlan, 1988; Running and Hunt, 1993).

The process of using BIOME-BGC required me to first perform a spin-up simulation, which slowly grew a simulated forest on the landscape until a dynamic equilibrium was met among climate, vegetation ecophysiology, soil organic matter (SOM), and nutrient pools (Thornton et al., 2002). The spin-up phase produced a restart file that described the state of the ecosystem and facilitated future runs of the model without re-establishing system equilibrium each run. Because land-use history (e.g. agriculture) affects the size of C and N pools above and below ground in present-day forest patches in the region, I altered the restart file to represent the agricultural land-use history of Dundee Township. I re-initialized C and N content in litter, live and dead forest stems, and coarse woody debris to 1% of their equilibrium value. I then initialized soil C and N in the fast microbial recycling pool to 1% and decreased the medium and slow microbial C and N pools by 30% and 15%, respectively, to reflect the alteration to soil as reported in the literature (Table 3-2). Then I was able to simulate forest stand growth from 1930 to 2006, which approximated the age of the forest site growing on prior agricultural land.

Typical calibration procedures involve comparing model output such as LAI and GPP to site-specific observations (e.g. Jung et al., 2007). I used chi-square statistics (χ^2) and altered two ecophysiological parameters to calibrate BIOME-BGC by assessing the goodness of fit between the study site observations of LAI ($3.36 \text{ m} \cdot \text{m}^{-2}$), above-ground carbon storage (9.13 kg C m^{-2}, average of values estimated in Table 3-1), and 1999 GPP ($759 \text{ g C} \cdot \text{m}^{-2}$) with the model output. I also wanted to restrain the number of parameter alterations of the default deciduous broadleaf biome, which has been established as generally representative of the eastern deciduous biome and undergone significant testing (White et al., 2000). Similar to Tatarinov and Cienciala (2006), I increased the rate of annual whole-plant mortality from 0.005 to 0.01, which is a 1% per year mortality rate. I increased the mortality rate to represent the increased mortality due to wind-throw, insect infestation, and disease that occur more frequently in fragmented and human-dominated landscapes. I also increased the fraction of leaf nitrogen in rubisco (parameter FLNR in the model) to 0.0361 from 0.033. A lack of data exists to accurately parameterize FLNR (White et al., 2000), but the generally accepted range for this value for eastern deciduous broadleaf forests in using BIOME-BGC is between 0.033 and 0.2 (William M. Jolly, University of Montana, personal communication; Galina Churkina, Max-Planck Institute for Biogeochemistry, personal communication). Altering these two parameters led to a χ^2 = 0.28, which showed no significant difference between the three observed metrics and the corresponding model output metrics.

Table 3-2: Existing literature describing carbon (C) and nitrogen (N) loss due to agricultural land use.

% C Loss	% N Loss	Sample N	Depth	Location	Forest Type	Study Type	# of Literature Sources	Source
43.4 (4.0)	-	14	A horizons	Global	All Types	Review	7	Davidson and Ackerman (1993)
36.8 (3.7)	-	14	A and B horizons	Global	All Types	Review	7	Davidson and Ackerman (1993)
31.5 (4.4)	-	18	A and B horizons	Global	All Types	Review	8	Davidson and Ackerman (1993)
14.7 (7.2)	-	21	Entire solum	Global	All Types	Review	5	Davidson and Ackerman (1993)
34.0 (4.4)	-	20	fixed depth - top layer	Global	All Types	Review	8	Davidson and Ackerman (1993)
26.2 (4.6)	-	25	fixed depths > 30 cm	Global	All Types	Review	9	Davidson and Ackerman (1993)
25.9 (3.6)	-	55	All data	Global	All Types	Review	18	Davidson and Ackerman (1993)
45.3 (5.9)	-	35	0 - 15 cm	Global	All Types	Review	-	Murty et al. (2002)
19.2 (2.6)	-	27	0 - 45 cm	Global	All Types	Review	-	Murty et al. (2002)
23.8 (3.0)	15.4 (3.5)	61	-	Global	All Types	Review	-	Murty et al. (2002)
27 (4.6)	15.8 (5.5)	29	-	Global	All Types	Review	-	Murty et al. (2002)
24	16	4	0 - [23.0 - 39.5] cm	Bond Head, Ontario	Red pine, white pine	Research	-	Ellert and Gregorich (1996)
20	-17	4	0 - [23.0 - 39.5] cm	C. Blondeau, Ontario	White pine	Research	-	Ellert and Gregorich (1996)
55	38	4	0 - [23.0 - 39.5] cm	Delhi, Ontario	Maple, beech, white oak	Research	-	Ellert and Gregorich (1996)
68	33	4	0 - [23.0 - 39.5] cm	Edwards, Ontario	Hemlock, white pine	Research	-	Ellert and Gregorich (1996)
38	12	4	0 - [23.0 - 39.5] cm	Exeter, Ontario	Oak, ironwood	Research	-	Ellert and Gregorich (1996)
23	-7	4	0 - [23.0 - 39.5] cm	Fonthill, Ontario	Red oak, white pine	Research	-	Ellert and Gregorich (1996)
8	-2	4	0 - [23.0 - 39.5] cm	Highgate, Ontario	Maple, beech	Research	-	Ellert and Gregorich (1996)
4	-7	2	0 - [23.0 - 39.5] cm	Kapuskasing, Ontario	Pine, black spruce	Research	-	Ellert and Gregorich (1996)
30	4	4	0 - [23.0 - 39.5] cm	Kemptville, Ontario	Pine	Research	-	Ellert and Gregorich (1996)
24	15	4	0 - [23.0 - 39.5] cm	Panmure, Ontario	Jack pine	Research	-	Ellert and Gregorich (1996)
40	26	4	0 - [23.0 - 39.5] cm	Plainfield, Ontario	Maple, hemlock, beech	Research	-	Ellert and Gregorich (1996)
34	27	4	0 - [23.0 - 39.5] cm	Ste. Anne, Ontario	Sugar maple	Research	-	Ellert and Gregorich (1996)
29	33	4	0 - [23.0 - 39.5] cm	Vineland, Ontario	Cheery orchard	Research	-	Ellert and Gregorich (1996)
49	36	8	0 - [23.0 - 39.5] cm	Winchester, Ontario	Maple, beech, white oak	Research	-	Ellert and Gregorich (1996)
47	43	4	0 - [23.0 - 39.5] cm	Woodslee, Ontario	Shagbark hickory	Research	-	Ellert and Gregorich (1996)
34	19	66	0 - [23.0 - 39.5] cm	All Sites	All Types	Research	-	Ellert and Gregorich (1996)
9	-	537	Various depths	Global	All Types	Review	74	Guo and Gifford (2002)
42	-	37	0 - 60 cm	Global	All Types	Review	-	Guo and Gifford (2002)
40	-	469	0 - 15 cm	Global	All Types	Review	50	Mann (1986)
26	-	212	15 - 30 cm	Global	All Types	Review	50	Mann (1986)
42	-	86	30 - 45 cm	Global	All Types	Review	50	Mann (1986)
26	-	274	0 - 30 cm	Global	All Types	Review	50	Mann (1986)
29	8	625	0 - 15 cm	U.S. Locations	All Types	Review	-	Post and Mann (1990)
22	4	625	0 - 30 cm	U.S. Locations	All Types	Review	-	Post and Mann (1990)
23	6	625	0 - 100 cm	U.S. Locations	All Types	Review	-	Post and Mann (1990)

- No available data reported, Standard Error reported in parentheses for % C loss where available.

In addition to calibrating BIOME-BGC to the study site, it was also necessary to parameterize the model using atmospheric and landscape data. I synthesized historical atmospheric carbon dioxide trends using measurements from the Law Dome DE08 and DEO8-2 ice cores (1930 - 1957 - Etheridge et al., 1998) and in situ air samples collected at Mauna Loa Observatory, Hawaii (1958 - 2006 - Keeling et al., 2005). Nitrogen deposition values were based on the assumption that nitrogen emissions are representative of deposition. I used literature on national air quality and pollutants (EPA, 2000; 2003; 2007), EDGAR-HYDE 1.4 (Van Aardenne et al. 2001 adjusted to Olivier and Berdowki 2001), and National Atmospheric Deposition Program grid data (NADP,

2008) to construct nitrogen deposition values from 1930 to 2006 using methods similar to Han (2007).

Daily climate data (i.e. daily precipitation and maximum and minimum temperature) were obtained from the National Climatic Data Center (NCDC). Nearest local station data (Ann Arbor, Dundee, Milan, Willis, and Ypsilanti, MI) were used to construct historical climate records from 1930 to 2006. I replaced NCDC observations with those obtained in the field for 2006 where available and used the MTCLIM model to produce daily values of short-wave radiation (W m^{-2}), vapor pressure deficit (Pascals), average daylight temperature, and day length. Together these seven climatic variables along with year and year-day provided the climatic input to BIOME-BGC.

Forest cover data for Dundee Township 2001 were obtained from the National Land Cover Database (NLCD, Homer et al., 2007) at a resolution of 30m. I reclassified the following NLCD land cover classes to forest: woody wetlands (1.9%, 2353.5 ha), mixed forest (0.15%, 182.7 ha), deciduous forest (8.4%, 10566.9 ha) and then resampled the data to a 15 m x 15 m cell size to correspond to my field measurements. From the resulting forest data layer, I created three grids that defined the distance each cell was from the east-, south-, and west-facing edge of the forest patch. Then I converted these edge orientation grids to a point vector file and spatially joined each edge-point file to obtain a single point file with the degree of edge for each cardinal direction for all forest grid cells in the township (a total of 58,400 points). Since BIOME-BGC also requires rooting depth, soil texture, elevation, and slope information for each location, I intersected the point layer with Soil Survey Geographic (SSURGO) soils data and Michigan Center for Geographic Information (MCGI) elevation and slope data. From SSURGO I acquired the rooting depth as the depth from the top of the A horizon to the bottom of the C horizon as well as soil texture characteristics (i.e. percent sand, silt, clay). Some of the NLCD-defined forest areas were removed from the analysis as they did not match soil delineations defined by SSURGO. For example, areas delineated as forest by NLCD and also as water, pits and quarries, and pits-aquents complex by SSURGO were removed from the analysis. This left 56,768 points remaining. The attribute table was then imported into XL-BGC, the Microsoft Excel version of BIOME-BGC. The equations, variables, and parameters in XL-BGC are virtually the same as in its Unix counterpart.

2.4 Computational Experiments

Four temperature treatments. I developed four air-temperature treatments to apply within each of two modeling experiments to determine the effects of climate and patch heterogeneity on carbon pool and flux estimates. In the first treatment, I applied locally gathered National Climate Data Center (NCDC) air-temperature data to all forest locations. This treatment provided a reference for comparison for other treatments because it is typically NCDC data or data obtained from similar methods that are used by BIOME-BGC users.

In the second treatment, I used the field-based air-temperature data (daily minimum, maximum, and daylight average) I collected external to the forest patch (year 2006). A correlation analysis between the external forest patch data and the reference data using a Pearson correlation coefficient statistic (0.859) and Levene's variance test ($p = 0.00$) showed that the external study patch and NCDC air-temperature data were significantly correlated and have similar variance at $\alpha = 0.05$. Given their high correlation, I applied the difference between field-based and reference air-temperature data from 2006 to the same year-day for historical data from 1930 to 2006.

The third treatment used the field-based air-temperature data I collected in the forest patch interior. A high correlation coefficient statistic (0.881) between the interior forest patch and NCDC air-temperature data was also found, which allowed me to use the same method as above to alter historical temperature values for the third treatment. I refer to each of these three treatments as *reference temperature*, *exterior temperature*, and *interior temperature*, respectively. In each of these three air-temperature treatments the temperature across the landscape was homogeneous in that all locations received the same temperature.

The fourth and last treatment was designed to be the best simulation of patch sizes, shapes, and edge effects in forest patches. Again, using the method described above, I altered the historical air-temperature data for each of the 5 edge zones measured for the east-, south-, and west-facing forest edge aspects as well as interior forest data (16 zones in total). In cases where an overlap existed between edge zones I applied a weighted mean of the temperature given by Eq. (2):

$$T_d = \frac{d_e * t_e + d_s * t_s + d_w * t_w}{d_e + d_s + d_w} \qquad (2)$$

where T_d is the edge distance weighted average temperature, d_e is the edge zone weight of the temperature measurement from the east-facing edge where temperature t_e was taken, and correspondingly d_s and d_w are similar values for temperature measurements at t_s and t_w, respectively. When the distance of a cell was beyond 60 m from the east-, south-, and west-facing edges I used temperature data obtained from the forest interior. Effectively, this last scenario implements a heterogeneous air temperature across the landscape that can differentially affect local carbon storage and flux. I refer to this treatment hereafter as the *heterogeneous temperature treatment*.

Real landscape experiment. In the first modelling experiment, I quantified the effects of using each of the four different air-temperature treatments on forest carbon storage in the highly fragmented Dundee Township, Southeastern Michigan, using actual land cover (15 m resolution). The landscape characteristics of Dundee Township are heterogeneous, therefore the results were derived by running BIOME-BGC for each of 56,768 forest point locations that collectively formed 262 forest patches or 10% of the area of Dundee Township. I then scaled these results to the conterminous United States to provide an indication of the degree of influence edge effects could have on national carbon storage estimates.

Hypothetical landscape experiment. In the second experiment, I sought to generalize the previous experiment results by reporting the sensitivity of carbon storage to variation in the spatial pattern of forest patches. Using hypothetical landscapes similar to those created by Franklin and Forman (1987), I created a ~992.25 ha (210 x 210 cell, 15 m resolution) landscape with 50% forest cover. I then altered the edge/area ratio metric of forest pattern to evaluate the effects of pattern on carbon storage. In each hypothetical landscape soil texture, slope, and elevation are homogeneous (Figure 3–2).

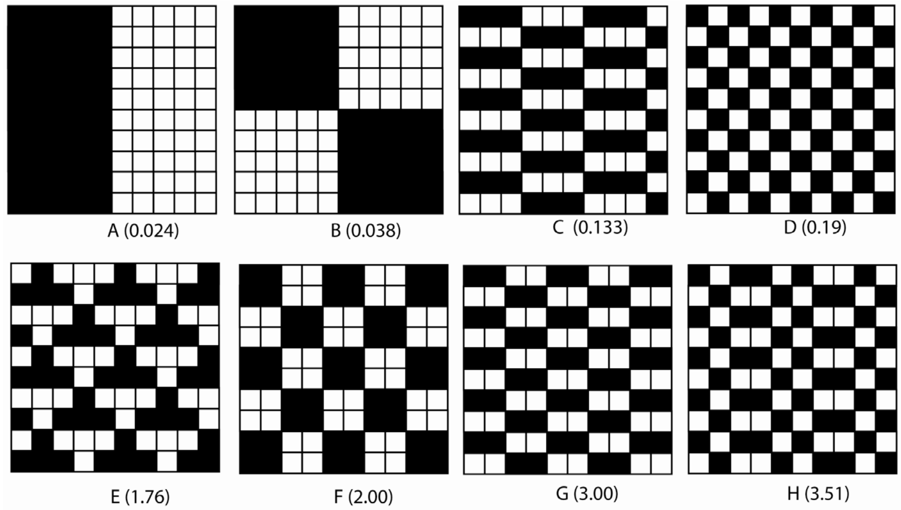

Figure 3–2: Hypothetical landscape patterns. Values in parentheses denote edge/area ratio for landscape with illustrated pattern. (A)-(D) depict complete landscapes with 50% forest. While all landscapes have a resolution of 15m, each cell in (A)-(D) represents a collection of 21 x 21 cells and each cell in (E)-(H) are the individual 15 m^2 cells. At the aggregate level the patterns in (E)-(H) were not visible and therefore I had to zoom in on the landscape. White squares denote areas of forest, while black squares denote areas of no forest. Additional landscapes not shown here include complete fragmentation, i.e., alteration of forest non-forest for every cell (4.00), 3 x 3 cell blocks (0.67), 4 x 4 cell blocks (0.52), and 7 x 7 cell blocks (0.29).

Each simulation within each experiment consisted of a number of forest patch assumptions. For instance, patches were assumed to be homogeneous in age, height and density. Patch edges were defined as open (Matlack, 1993) and assumed to be adjacent to open fields or residential turf. For each simulation I report measurements of vegetation C, litter C, soil C, and total C at the end of 2006. Cell resolution in each experiment was 15 m (225 m^2) and the total area represented was 992.25 and 12587 ha for the hypothetical and Dundee Township landscapes, respectively. I sum the results across all cells in the landscape for each treatment, and therefore the results report the aggregate outcome of each hypothetical landscape and Dundee Township after 77 years of forest growth.

3. RESULTS

3.1 ANALYSIS OF FIELD DATA

The estimated median temperature from a non-parametric Friedman test revealed a decrease in maximum temperature from the forest patch edge to a depth of 80m (Figure 3–3). Compared against critical χ^2 values at $\alpha = 0.01$ the computed χ^2 values (c.v. = 13.28 for each transect and 30.58 for all sensors) were substantially greater and provide evidence that temperature among measurement locations were significantly different within transects and among all sensors. Median ranks for the minimum temperatures are less prominent (Figure 3–3) and therefore I focus the remainder of the discussion on maximum temperature.

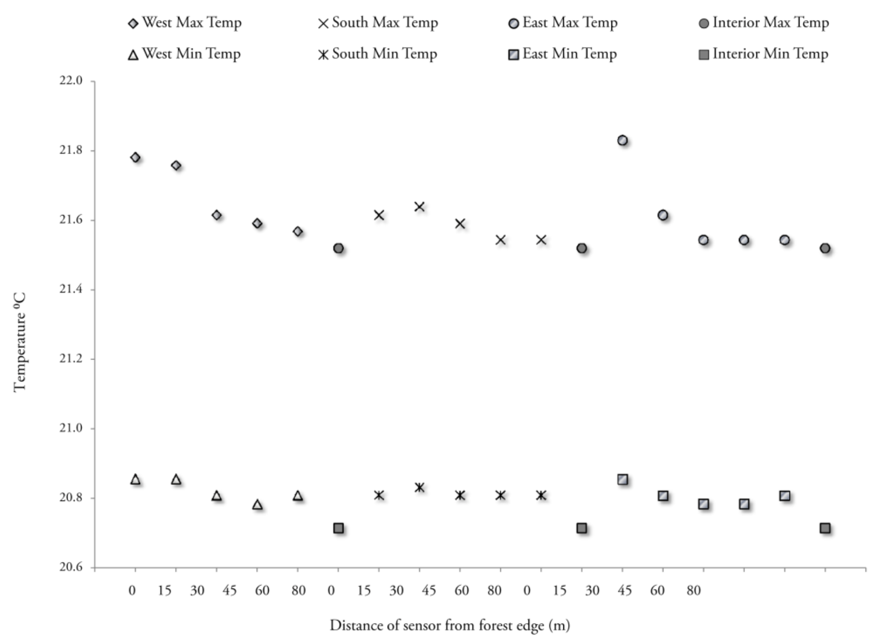

Figure 3–3: Median values of maximum and minimum temperature for each sensor by distance from forest edge.

The functional data analysis (described in Section 2.2) revealed an overall trend of decreasing maximum temperature from the forest edge inward. During daylight hours this trend existed for approximately 83%, 80%, and 94% of the measurements on the west, south, and east edges, respectively (Table 3-3).

Table 3-3: Summary statistics of functional analysis demonstrating the temperature gradient direction from forest patch edge to the patch interior. Negative slope (m) = cooler towards the interior, conversely m > 0 means it is warmer towards the interior; m = 0 means no gradient. T_{obs} = maximum temperature values observed.

	All T_{obs}			6 am > T_{obs} > 6 pm			6 am < T_{obs} < 6 pm		
	West	South	East	West	South	East	West	South	East
$m > 0$	28.17%	35.02%	33.97%	40.79%	55.68%	62.42%	15.54%	17.53%	5.53%
$m = 0$	4.21%	4.45%	5.29%	7.05%	7.26%	9.86%	1.36%	2.14%	0.72%
$m < 0$	67.59%	60.50%	60.66%	52.08%	37.06%	27.56%	82.93%	80.33%	93.67%

In addition to illustrating the existence of air-temperature alterations from the forest patch edge to the patch interior (Figure 3-4), results show a switch near sunrise (i.e. ~6 a.m.) from a warmer interior and cooler exterior to the reverse. The east edge experienced a much greater warming in the morning when low sun angles were directed on the eastern patch edge. The warming of the east edge decreased near noon as the sun passed overhead, which then began to warm the western edge. The south-facing edge to interior air-temperature gradient was lower (i.e. less slope) but experienced increased persistence of edge effects from sunrise to sunset (data not shown).

3.2 Analysis of Dundee Township carbon storage responses

Using BIOME-BGC I produced simulated measures of three carbon pools (kg C m^{-2}): vegetation, litter, and soil carbon. Vegetation C includes leaf, live coarse and fine root, live and dead stem carbon; litter carbon includes coarse woody debris (CWD) and litter carbon found in labile, unshielded cellulose, shielded cellulose, and lignin litter pools; soil carbon is the sum of carbon found in fast, medium, and slow microbial recycling pools in the mineral soil as well as that in recalcitrant soil organic matter (SOM) pool. The sum of the three pools provides a measure of total ecosystem carbon (Figure 3-5).

Results from applying each of the four air-temperature treatments indicated a lower level of soil carbon for the reference treatment compared to the other treatments (Figure 3-5). On average temperature values ranked from coolest to warmest were: reference, interior, heterogeneous, and exterior with the mean daily temperature value

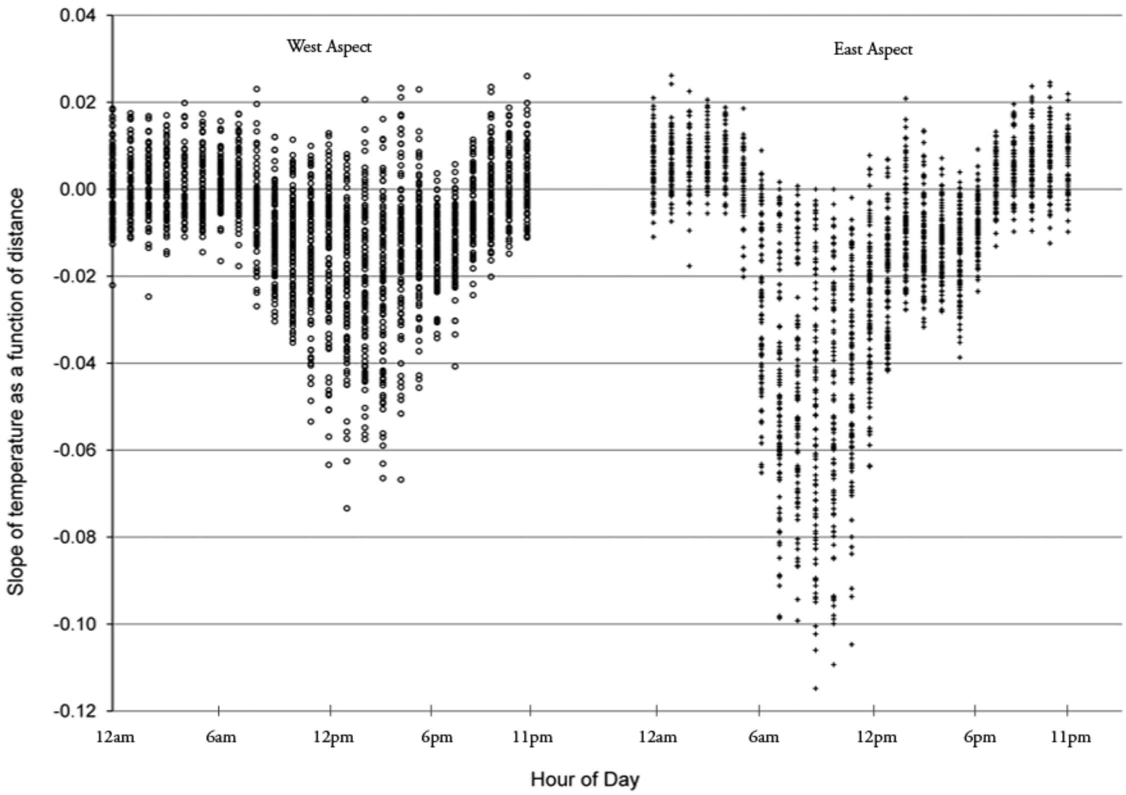

Figure 3–4: Hourly slope values calculated from repeated linear regressions of temperature by distance for east and west forest patch aspects. Plotted values (slope) at zero indicate no edge effect, while values greater than zero demonstrate a warming trend from the patch edge to the patch interior and negative values indicate a cooling gradient from patch edge to the patch interior.

(˚C) of each treatment over the 77 years as 9.20, 9.46, 9.65, and 10.02, respectively. Estimated levels of soil C followed: 127 437, 140 321, 141 903, 148 422 Mg C respectively. Therefore, growing forest across Dundee Township for 77 years under a 0.82˚C range of air temperature led to a difference of up to 20 985 Mg of soil carbon in the aggregate or 1.64 kg C · m^{-2} of forest area. Assuming that the heterogeneous treatment provides the most realistic account of air-temperature conditions, relative to the output of soil C from the heterogeneous treatment, the interior treatment underestimated soil C by 1.24 Mg per ha and the exterior treatment overestimated soil carbon by 5.1 Mg per ha.

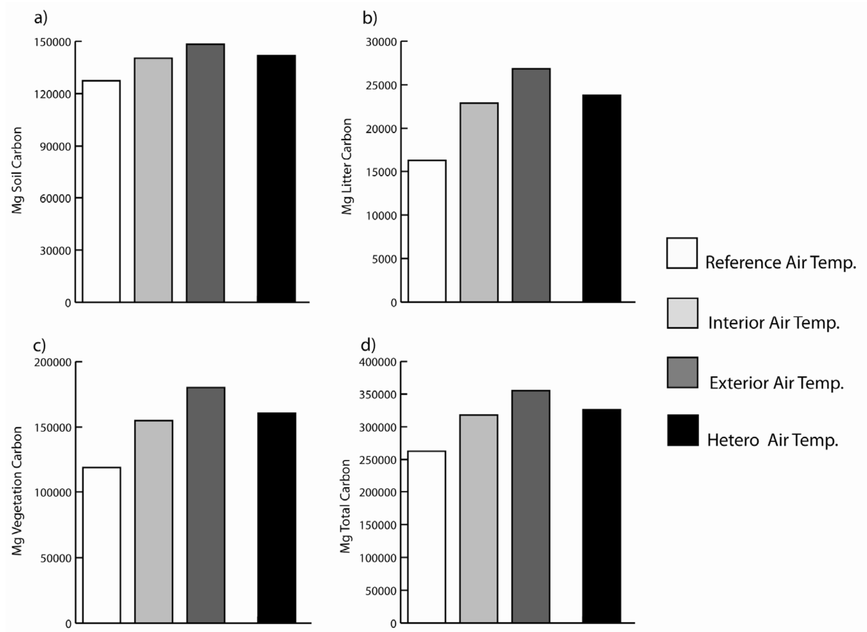

Figure 3–5: Homogeneous vs. heterogeneous air-temperature treatments. Values represent total amount of carbon in soil (a), litter (b), vegetation (c), and all pools in total (d), as simulated, for all forest patches summed across Dundee Township under the real landscape experiment. Each of the Reference (climate data provided by the National Climate Data Center, NCDC), Interior (use of field data obtained within the interior of the forest patch), and Exterior (use of microclimate data obtained external to the forest patch) air-temperature treatments are homogeneous in that the same climate was applied to all forested area in the township. The heterogeneous temperature treatment (Hetero Air Temp.) used field data obtained along transects from the forest patch edge to the interior on the east-, south-, and west-facing aspects.

Litter carbon, the smallest of the three carbon pools measured, experienced a range of 10 Mg of carbon difference among the air-temperature treatments. Again the exterior treatment produced the highest levels of litter carbon at 26 836 Mg for the entire township, followed by the heterogeneous treatment with 23 799 Mg, the interior treatment (22 892 Mg), and the reference treatment at 16 271 Mg C. Differences in litter carbon directly resulted from differences in vegetation carbon, which showed the same treatment ranking from highest to lowest.

Vegetation carbon pools were slightly larger than soil carbon pools, which is indicative of the distribution of above and below ground C in eastern deciduous forests near 40˚ latitude (Dixon et al., 1994). Vegetation C accounted for the majority of the carbon discrepancy among treatments with an overall range 61 280 Mg C among treatments. Values of 118 888, 160 570, 180 168, and 154 866 for each of the reference, heterogeneous, exterior, and interior treatments, respectively, illustrate the large difference between representing within-patch heterogeneity (i.e. ~6000 Mg C underestimation using interior treatment relative to the heterogeneous treatment and ~20 000 Mg C overestimation using the exterior treatment) versus homogeneous patch measurements. These vegetation C outputs corroborate theory and empirical work that illustrate the positive influence of increased temperature on vegetation growth when all other factors, such as soil water and nutrients, are not limiting (Lieth, 1975; Schlesinger 1997). The difference in vegetation C was much larger than either of the soil or litter C pools, indicating that vegetation production in BIOME-BGC is much more sensitive to air temperature differences than is decomposition of soil carbon. The range of treatments led to a relative difference vegetation C of up to 4.80 kg C · m^{-2} in forested areas.

Total estimated carbon storage for Dundee Township forest was 262 596, 318 079, 326 272, 355 426 Mg C for each of the reference, interior, heterogeneous, and exterior air-temperature treatments. The larger estimates from field-based temperature treatments (55 483 - 92 830 Mg more) compared to that obtained by the reference treatment can be explained by increased vegetation growth due to higher the temperatures of the field-based treatments. The increased growth positively feeds back onto increased litter and soil C pools, which further facilitate vegetation growth. These results illustrate the influence of landscape heterogeneity on model results such that differences in temperature taken at a field study site versus some distance away may lead to discrepancies at the township level of 92 830 Mg C or 7.27 kg C · m^{-2} in forested areas.

Estimates of carbon storage in each pool are based on ~56 000 independent simulations that make up the forest cover in Dundee Township as illustrated in Figure 3-6. While heterogeneous site characteristics did alter total C values across the township, because the reference, interior, and exterior treatments applied a homogeneous climate to the township, it was often the case that individual patches produced very similar levels of total carbon storage. In contrast the heterogeneous treatment produced varied carbon levels within the patch and among patches.

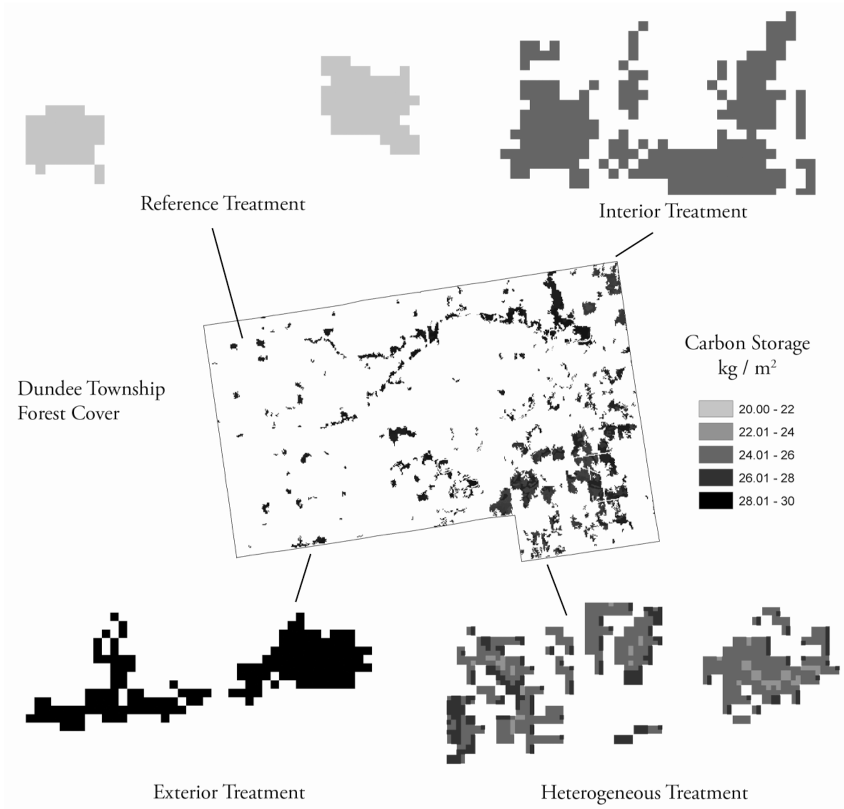

Figure 3–6: Spatial distribution of deciduous forest in Dundee Township (center) and typical carbon storage output for different forest patches under the following four climate data treatments: reference (upper left), interior (upper right), exterior (lower left), and heterogeneous (lower right). Treatments are as described in previous figures and in Section 2.3 model simulations.

3.3 Analysis of carbon storage response to landscape fragmentation

Application of the 3 homogeneous air-temperature treatments to 12 hypothetical landscapes (Figure 3-2), each with 50% forest but different edge/area ratio patterns, produced results similar to the Dundee Township experiment. Specifically, the largest estimate of carbon storage occurred using the exterior air temperature treatment (13 986 Mg C) followed by the interior and reference treatments, 12 350 Mg C and 10 093 Mg C respectively (Table 3-4). Because I held the site conditions constant in this experiment, all 12 landscape patterns produced the same result in each of the three homogeneous air-temperature treatments.

Table 3-4: Description of hypothetical fragmented landscape characteristics and associated carbon storage estimates. Cell edges and areas represent summed values across the hypothetical landscape.

Air Temperature Treatment	Landscape Description	Figure 2	# Cell Edges	Forest Area (# Cells)	Edge/Area (Cells)	Edge/Area (m/m2)	Vegetation (Mg C)	Litter (Mg C)	Soil (Mg C)	Total (Mg C)
Heterogeneous	Parallel patches	A	630	22050	0.03	0.002	6252.88	870.86	5269.55	12393.29
	2 large square patches	B	920	22050	0.04	0.003	6248.74	870.10	5267.63	12386.47
	Large horizontal patches	C	2936	22050	0.13	0.009	6264.53	871.48	5275.35	12411.36
	10 x 10 cell patches	D	4200	22050	0.19	0.013	6356.31	886.11	5296.96	12539.38
	7 x 7 cell patches	-	6300	22050	0.29	0.019	6607.36	926.51	5375.55	12909.43
	4 x 4 cell patches	-	11440	22050	0.52	0.035	6741.99	950.71	5429.52	13122.22
	3 x 3 cell patches	-	14700	22050	0.67	0.044	6857.59	971.11	5477.23	13305.94
	T shaped patches	E	38723	22050	1.76	0.117	6906.11	980.84	5501.57	13388.52
	2 x 2 cell patches	F	44102	22050	2.00	0.133	6925.70	982.92	5504.64	13413.27
	2 cell horizontal patches	G	66150	22050	3.00	0.200	6891.61	978.04	5494.33	13363.97
	-	H	77282	22050	3.50	0.234	6974.89	994.11	5533.89	13502.89
	Complete checkerboard	-	88200	22050	4.00	0.267	7056.60	1009.89	5572.70	13639.18
Exterior	All Landscapes Equal	-	-	-	-	-	7321.32	1033.35	5631.28	13985.95
Interior	All Landscapes Equal	-	-	-	-	-	6222.62	866.32	5261.06	12350.00
NCDC	All Landscapes Equal	-	-	-	-	-	4759.11	607.26	4726.57	10092.95

In contrast to the homogeneous air-temperature treatments, the heterogeneous treatment results depended on the degree of fragmentation in the landscape. Specifically, as the landscape was altered from a single large patch to complete fragmentation (i.e. a checkerboard), carbon storage increased logarithmically to approximate the function $y = 268.13\ln(x) + 13\ 204$, ($r^2 = 0.92$), where x is the ratio of total cell edge / total cell area across each hypothetical landscape. The logarithmic growth in carbon storage moved carbon estimates that closely aligned with the interior air-temperature treatment, when no fragmentation was present in the landscape, toward the exterior air-temperature treatment values when forest cover was completely fragmented (Figure 3-7). Total carbon storage values show that fragmentation of a relatively unfragmented landscape (edge/area from 0 to 0.67) had more than three times the effect on carbon storage than did further fragmentation of a highly fragmented landscape (edge/area from 0.67 to 4; Table 3-4).

4. Discussion

4.1 Carbon responses

By back-casting the difference between each of my 2006 field-based treatments and the 2006 reference treatment to the years extending from 1930 to2006 and simulating carbon storage values across Dundee Township I was able to evaluate the effects of within-patch and landscape heterogeneity in air temperature on carbon storage in a fragmented and human-dominated landscape. Specifically the interior, heterogeneous, and exterior treatments produced 318 080, 326 272, and 355 427 Mg C, respectively, in

Dundee Township at the end of 2006. Therefore, within-patch air-temperature heterogeneity produced an 8000 Mg C difference from the interior treatment and nearly a 30 000 Mg C difference from the exterior treatment. Clearly the choice and application of field-based temperature measurements is an important decision given that the overall range of outcomes differed by ~38 000 Mg C, and if we include the influence of landscape heterogeneity by adding the results from the reference treatment extends the range of uncertainty to ~93 000 Mg C among treatments.

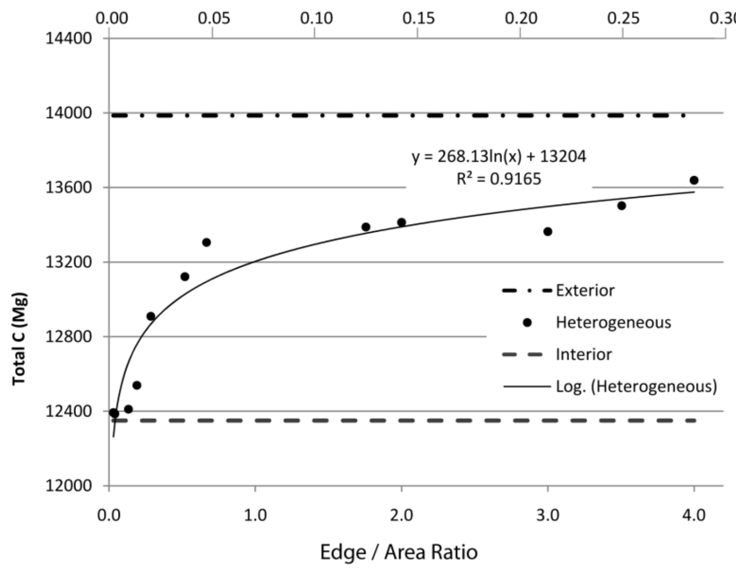

Figure 3–7: Logarithmic increase in total carbon storage with landscape fragmentation as measured using the 12 hypothetical landscapes illustrated in Figure 3–2 and described in Table 4. The upper x-axis displays edge/area ratio in units of meters (i.e. edge in meters / area in m^2) while the lower x-axis displays edge/area ratio in cell units (i.e. number of cell edges / number of cells). The logarithmic function in meters is y = 267.7ln(x) + 13929 (r^2 = 0.92).

The range of uncertainty in my carbon estimates at the landscape scale is influenced by the relative difference among air-temperature measurements. Published climate data from nearby stations (i.e. the reference treatment) had the lowest maximum temperature compared to field-based measurements outside and inside the forest patch (i.e. exterior and interior treatments) in 75% and 32% of the measurements, respectively, out of the 228-day range of collected data. In the initial years of model output, results showed a higher level of estimated C accumulation in soil and litter for the reference treatment relative to the other treatments, corroborating theory that cool moist temperatures harbor increased soil C relative to warmer and dryer climates (Schlesinger, 1997). However, because the interior, heterogeneous, and exterior temperatures were greater than the reference data, and because in BIOME-BGC the vegetation C pools proved to be more responsive (in positive C aggradation) than the soil C pools (in reduced C accumulation due to increased decomposition), the overall carbon storage was larger for each of those three treatments in both experiments when compared to carbon storage values for the reference treatment. The positive feedbacks of biogeochemical and nutrient cycling continued to differentiate the treatments over time due to the different external climate forcings imposed by the temperature treatments.

The temperature treatments contained various forms of heterogeneity and corresponding degrees of realism. While reference data were obtained from sources typically used by ecosystem modelers (e.g. National Weather Service stations), station measurements can be poorly sited such that they are influenced by shadowing, humidity, isolation from wind, and other factors (Davey and Pielke Sr., 2005; Peterson, 2006). Furthermore, station measurements can be inhomogeneous over time and require adjustment due to station relocation, equipment change, and land-use and land-cover change (Karl and Williams Jr., 1987; Pielke Sr. et al., 2007b). Post-experiment investigation of why the reference treatment held lower values than the field-based measurements revealed that the Dundee NWS station was located in the middle of a wastewater treatment plant. Not only was the station surrounded by open liquid waste treatment pools but also the treatment plant was within proximity to a river on three sides. The result was a station location that experienced a very local microclimate that had substantially greater humidity then locations less than half a kilometer away. Assuming all climate and radiation conditions are equal, increased humidity will lead to a reduction in the surface air temperature (Pielke Sr. et al., 2007a), explaining why the reference treatment had lower temperature values.

Using hypothetical landscapes, similar to those developed by Franklin and Foreman (1987) to explore the general ecological consequences of fragmentation and by Smithwick et al. (2003) who investigated the effects of wind and light on forest patch carbon storage, I evaluated the effects of within-patch air-temperature heterogeneity on carbon storage in a number of fragmented landscapes. The logarithmic response of carbon storage to fragmentation suggests that under low fragmented landscapes interior forest air-temperature measurements may provide a closer approximation of air-temperature conditions within large forest patches than measurements taken exterior to the patch or typically acquired from NCDC or National Weather Service Stations. Conversely, in a highly fragmented landscape, air-temperature measurements taken exterior to the forest are more representative than those taken interior (Figure 3-7).

While these results corroborate theory and empirical work that demonstrate increased productivity with increased temperature (Lieth, 1975; Schlesinger 1997), this modelling analysis does not take into account the effects of the following on carbon storage along or within edge zones: mortality and carbon removal along edge (e.g. by adjacent streams; Malanson and Kupfer, 1993), wind throw and light penetration (Smithwick et al., 2003), harvest and fire disturbance regimes (Smithwick et al., 2007), varying species composition or invasive species in edge habitats, or soil organic carbon changes due to land use and erosion (Yadav and Malanson, 2008). These processes could alter the presented results and are important topics for future research, but are beyond the scope of this paper.

4.2 Does the incorporation of edge effects influence forest patch carbon storage?

Analysis of both real and hypothetical fragmented landscapes showed that within the range of observed temperature values and their corresponding MTCLIM climatic variable values, the BIOME-BGC carbon cycling processes and forest ecosystem carbon budgets were highly sensitive to edge effects. Under the real landscape experiment, the heterogeneous treatment (i.e. including edge effects) produced an overall ecosystem carbon storage value of 326 272 Mg C or an average of 25.55 kg C · m^{-2}. Under the same experiment, using the less realistic homogeneous climate treatments, the interior and exterior treatments produced 318 080 Mg C (24.91 kg C · m^{-2} on average) and 355 427 Mg C (27.83 kg C · m^{-2} on average). Therefore basing carbon storage assessments on interior forest microclimate measurements could underestimate values by 0.64 kg C · m^{-2} or ~2.5%. More likely is the case that researchers using ecosystem biogeochemistry models like BIOME-BGC use climate measurements made in nearby open fields or at meteorological stations, in which case the analysis shows a large overestimation of 2.28 kg C · m^{-2}, or ~8%, on average.

These results are significant because attempts to account for carbon uptake and storage in temperate forests have been necessary as part of the ongoing and broader effort to (a) close the carbon budget and account for the temperate carbon sink, and to (b) include more realistic representations of the effects of people on landscapes into ecosystem process models. By scaling up the average C storage values to the continental United States I can obtain an idea of the relative effect that within-patch heterogeneity could have on national carbon accounting. For example, using the 2001 National Land Cover Dataset (Homer et al., 2007) I found that the interior, heterogeneous, and exterior treatments produced 23.15, 23.75, and 25.89 Pg of total carbon storage, respectively, for deciduous forests across the United States. Despite the obvious and important simplifications involved in this scaling exercise, these results suggest that choosing a homogeneous representation of forest microclimate could underestimate carbon values by 0.6 Pg or ~2.6% when interior forest measurements are used or overestimate carbon storage values by 2.14 Pg or ~8% when temperature measurements are collected exterior to the forest patch. The range of over- and under-estimates pose a serious question about the degree to which within-patch heterogeneities influence carbon storage estimates since these differences among the modelling treatments are on the order of current estimates of the North American carbon sink (~1.7 Pg C yr^{-1} in the 1990s, Fan et al., 1998; ~0.65 Pg C yr^{-1} from 2000 to 2005, Peters et al. 2007; ~0.505 Pg, CCSP 2007). Clearly within-patch heterogeneity at the micro level has an influence on the macro-level patterns of carbon storage across the United States. How we go about collecting data on within-patch heterogeneity at national and global scales and incorporating those data into national and global estimates of carbon storage remains a significant future endeavor.

Another significant feature of these results is their implication for the use of vegetation process models to couple to regional or global climate models. I found that including patch heterogeneity tended to increase C accumulation and storage in forest patches as a result of increased forest production as simulated by BIOME-BGC. If patch heterogeneity does drive higher forest production per unit area, this would be reflected in larger leaf area, altering the canopy reflectance of short-wave radiation and the transfer of water vapour to the atmosphere through evapotranspiration.

4.3 Ecosystem processes in land-change models

By loosely integrating BIOME-BGC with a commercial GIS package I was able to (1) facilitate the retrieval of site characteristic data for running the model, and (2) visualize results of ecosystem function metrics spatially. A tight integration would provide the capability to apply BIOME-BGC at a high resolution and in a highly fragmented environment where multiple runs of the model may be conducted to alter the detail of landscape data (i.e. reducing the number of runs by running the model on polygon attributes or the converse by running the model on more detailed point data) or to alter the landscape to assess land-use and land-cover change scenarios. Contemporary research coupling human and environment systems for integrated assessments of land-use and land-cover change (LUCC) have linked agent-based models (ABMs) and GIS to evaluate the effects of land-use policies on forest cover (e.g. Robinson and Brown, in press). The combined use of a dynamic agent-based approach with ecosystem process models can help us better understand (a) the coupled human-environment processes that influence ecosystem function in fragmented and human-dominated, (b) the effects current and future landscape trends may have on ecosystem function and productivity as well as surface-atmosphere coupling, (c) the effects of policy and management decisions on landscape structure and function have and could have on ecosystem function, and (d) how changes and knowledge of ecosystem function and services could feedback to influence land-use and land-cover change decisions and trajectories.

5. Conclusions and future research

Simulation of ecosystem carbon cycling using the widely used biogeochemical process model BIOME-BGC in a heterogeneous, fragmented, and human-dominated landscape, combined with field-measured microclimate temperature patterns, showed that within-patch and landscape level heterogeneity and landscape fragmentation had a strong effect on ecosystem simulated carbon storage in the landscape. Simulated forest productivity and carbon storage were underestimated when temperature data from a National Weather Service maintained station were used and overestimated when field measured temperature data were used, in comparison to the more realistic simulations of the fragmented landscape using microclimatic temperature differences.

The combined field and modelling study clearly demonstrated that simulated values for carbon cycling and ecosystem carbon stocks are highly sensitive to realistic patterns of forest fragmentation. Therefore as we improve our carbon estimating procedures, we need to include measurements of both within-patch and landscape heterogeneity and make strides to also incorporate human behaviors through land-use and land-cover change. Future research that identifies all permutations of fragmentation associated with different edge/area ratios may be able to provide insight for the creation of ranges of uncertainty associated with past and future carbon estimates. Similarly, statistics or indices of landscape heterogeneity may also prove to be useful in this endeavor.

The integration of biophysical models with land-use models allows for the identification of processes causing critical changes in ecosystem functions and the landscape patterns that act at different scales and are linked across scale. By quantifying the influence of human systems on ecosystems and how those ecosystems feedback we can better define (1) the impacts of humans on ecological systems, (2) the resilience and robustness of ecological systems to human-induced perturbations, and (3) the thresholds that result in ecosystem failure or decrease ecosystem function beyond their use to humans (Moran et al., 2005). Furthermore, if coupled with LUCC models of anthropogenic behavior, the representation of ecological outcomes by ecosystem process models can be used to quantify the effects of anthropogenic systems on the quality and functions of ecosystems (land-cover) and how those changes feedback to alter anthropogenic behavior. A tight integration of LUCC and ecosystem modeling approaches is the focus of novel research in these fields (Yadav et al., 2008).

6. Acknowledgements

Support for the presented research in the form of grants was provided by the National Science and Engineering Research Council of Canada (NSERC), the Graham Environmental Sustainability Institute (GESI) at the University of Michigan, and the National Science Foundation Biocomplexity in the Environment Program (BCS-0119804 and GEO-0814542) for Project SLUCE. I would also like to acknowledge with gratitude the intellectual support of my colleagues Leslie Sefsick, John S. Kimball, Rick Riolo, Scott Page, Donald R. Zak, Jason Taylor, Jose Garcia, Dan Bamber, Leslie Garrison, Greg Jacobs, Andrew Bell, Tingting Zhao, Haejin (Jinny) Han, Neil Carter, Jonathan Redmond, and Peter Johnson. Lastly, I would like to thank my coauthors and the anonymous reviewers and journal editor for their constructive comments that have led to the acceptance for publication of this paper in Ecological Modelling.

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Chapter 4 PARCEL SIZES AS DETERMINANTS OF LAND-COVER CHARACTERISTICS IN AN EXURBAN LANDSCAPE.

1. INTRODUCTION

Anthropogenic drivers of land-use and land-cover change (LUCC) have altered over 30% of the Earth's surface (Houghton 1994, Vitousek et al. 1997). Specifically, these surface changes alter biogeopyhsical (e.g. albedo, evaporation, and heat flux), biogeochemical (e.g. carbon and nutrient cycling), and biogeographical (e.g. location and movement of species) processes that in turn affect local and global climate (Bonan 1997, Stohlgren 1998, Kalnay and Cal 2003). While the relationship between surface alterations as a direct forcing on climate remains poorly understood, it has been estimated that they may be as important as greenhouse gases (Pielke 2005). That being said, estimates of CO_2 efflux to the atmosphere by LUCC approximates 30% of anthropogenic totals, which would make it the second largest contributor to global climate change after fossil fuel burning (Sarmiento and Sundquist 1992, Sundqist 1993). Because of the profound effect of LUCC on CO_2 emissions to the atmosphere, extensive research has focused on quantifying and mapping LUCC at coarse resolutions and global scales (e.g. Hansen et al. 2000).

Understanding the relationship between LUCC and global climate becomes difficult because LUCC is driven by local complexities and feedbacks in human behavior that are non-stationary across space and time (Foley et al., 2005). Similarly, heterogeneity in human characteristics and behaviors, biophysical characteristics of the landscape (e.g. soils or topography), and climate exist and have differential impacts on LUCC outcomes and trajectories. Incorporating micro-level heterogeneity warrants a bottom-up approach to LUCC research that focuses on 1) human perceptions of their environment and its management, 2) location accessibility, 3) distance to social, occupational, academic, and service amenities, 4) neighborhood similarity, and 5) government policy constraints among other drivers not mentioned.

Within LUCC research, modeling, understanding, and predicting urban growth has been a primary focus. The emphasis on urban growth research is partly due to rapid urbanization across the Earth. As of 2002, 47% of the human population lived within urban areas and it is estimated to increase to 65% by 2050 (UNEP 2002). While the rate of urbanization is alarming, definitions of urban area or urban land-cover in global reports remain ambiguous. Perhaps more striking than the rate of urbanization is the rate of low density development just beyond the urban fringe in developed countries and particularly in the United States. Classified as exurban areas and defined as having a housing unit density between 0.68 and 16.18 ha, in the United States, these areas are growing at a rate approximately 25% greater than the population and consuming over seven times the amount of residential land-use in urban and suburban areas (Theobald 2005). This dispersion of human population around the urban/suburban periphery is often overlooked, and the complex land-cover composition of exurban residential parcels complicate generalizations about the ecological effects of these types of land-use conversions.

Despite the magnitude of exurban growth and the extensive literature on land-use and land-cover change analyses, few researchers have analyzed LUCC data at resolutions finer than the county or urban area (Irwin and Bockstael 2007). As a step toward filling this lacuna, I describe how strongly land-cover amounts, variability, and patterns are related to residential parcel size and neighboring land-covers. This research addresses two specific questions through the analysis of spatial land-cover data. The primary question is: What is the distribution of quantity and pattern of land-cover types in response to residential parcel size, in the exurban landscape of Southeastern Michigan? In order to address this question I used a number of pattern metrics (i.e. total land cover, number of patches, mean patch size, edge density, and area weighted mean shape index). The secondary question is: what is the degree of autocorrelation of land-cover quantity among residential parcels?

2. Materials and methods

2.1 Study area

I focus the analysis on three townships, Pittsfield, Ray, and Scio Townships, in Southeastern Michigan (Figure 4–1). Collectively the townships represent a mixture of population growth rates, land uses, and land covers (Figure 4–2). While each township was primarily agrarian in 1950, Ray Township had a small population of 1671 individuals and grew slowly over time to remain predominately rural in 2000. Unlike Ray Township, both Pittsfield and Scio Township border the city of Ann Arbor, which has experienced high levels of growth. To the south of Ann Arbor, the already well populated Pittsfield Township grew exponentially, forcing substantial changes in land use and land cover. The typical land-use and land-cover transitions from 1950 to 2000 involved the removal of agriculture for residential use and farm abandonment (i.e. crops to tree cover). I have not plotted open natural areas such as savanna or grassland landscapes because they follow a very similar trend to that of tree cover. The same population and land-use and land-cover change trends occurred to a lesser degree in Scio Township, located just west of Ann Arbor.

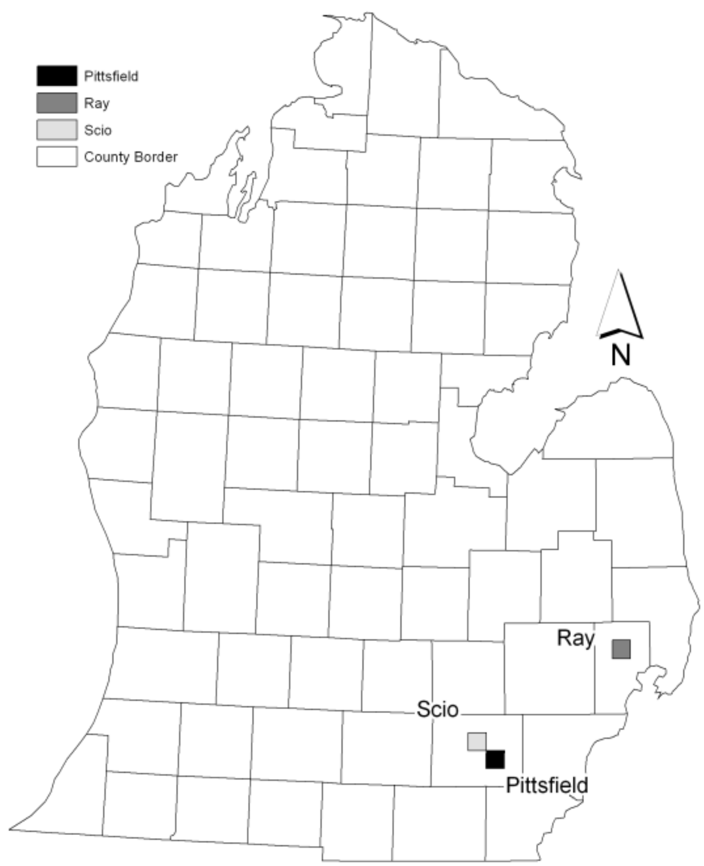

Figure 4–1: Location of the study townships in Southeastern Michigan, USA.

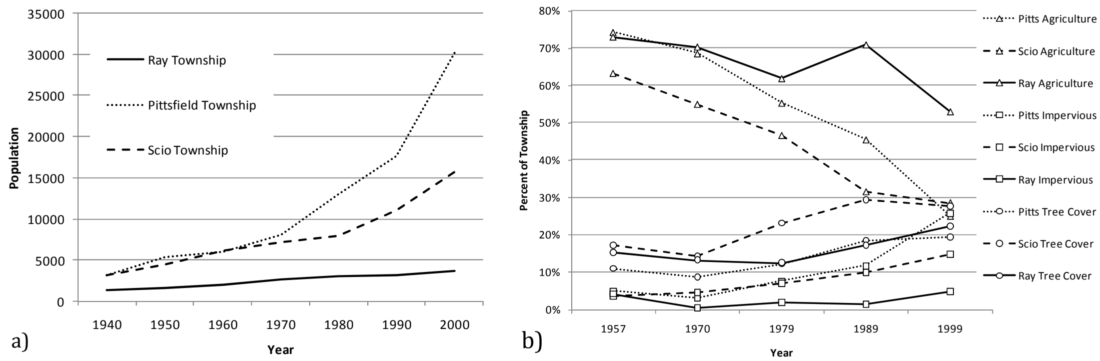

Figure 4–2: Trends in (a) population and (b) land-covers types for the three study townships. Population data obtained from U.S. Bureau of Census (1950, 1970, 2000). Land-cover data were derived from aerial photograph interpretation at the University of Michigan.

2.2 LAND COVER DATA

Land-cover data were manually digitized from aerial photograph interpretation at multiple dates for each township. On-screen digitizing was performed at a scale of 1:3000 with a linear minimum mapping unit (MMU) of 10 m. The MMU was used for all landscape features (e.g. trees and forest, water, and other natural patches) with the following exceptions. The strong linear boundaries of impervious surfaces (e.g. roadways) increased the ability to delineate edges of impervious land cover. Similarly, linear clusters of trees that were < 10 m in one dimension but greater in another were digitized to capture the tree cover patch.

Altogether, seven land-cover classes were digitized. Tree cover, consisted of all tree species, forest patches, and individual trees that exceeded the MMU. Cropland consisted of row crops, and other indiscernible natural areas that appeared to be of an agrarian nature. Open fields that did not show cropping striations or had a park-like appearance were classified as other natural areas (e.g. prairie). Maintained lawn consisted of manicured areas adjacent to commercial, industrial, and residential structures as well as golf courses and other maintained areas (e.g. recreation fields). Impervious lands consisted of 1) impervious structures (i.e. building footprints, driveways, and parking lots) and 2) impervious transportation (i.e. all impervious transportation network features). Wetland areas were saturated areas that were not 1) open water, 2) wooded wetlands, or 3) the result of recent precipitation events (e.g. flooded farm fields). Open water consisted of ponds, lakes, rivers and other hydrologic features. In a very limited number of cases land cover was in transition to development. In these cases I identified the land cover at its previous state if development or transition had just begun or as its future state if development had progressed to the extent that new road or structure outlines were visible.

The digitizing of aerial photos started with the most recent date (i.e. near year 2000). The most recent photos were often in color, had the least amount of splicing errors, and had the least amount of location error based on the orthorectification procedure, therefore I felt they offered the most accurate representation of the landscape for digitizing. Once all land covers were digitized for the most recent date, that data layer was copied to provide an accurate template for digitizing land cover from the previous date for which I had access to aerial photo data. Digitizing was then conducted on the template for year 1990 using polygon cut, merge, and delete functions. Often the procedure involved removing impervious structures or adding or removing tree cover and cropland. Effectively what this did was fix the location of the land cover data such that one could delineate an area (i.e. parcel) and obtain an accurate temporal description of land-cover change for that area. This procedure also required that on occasion I shift earlier aerial photo data to match the original 2000 year data.

While the on-screen digitizing was labor intensive, it arguably provides the most accurate representation of aerial photograph data (Kunapo et al. 2005) and, at a minimum mapping unit of 10 m, the resolution is finer than that available by other sources (e.g. NLCD, Homer et al. 2007).

2.3 Parcel data

Parcel size is an important determinant of land-cover because 1) the parcel is a legal administrative unit within which the environment is managed by the private land-owner, 2) parcel size is often the outcome of minimum lot-size zoning policies (Drzyzga and Brown 2002), a constraint imposed by local land-use policies to achieve some socio-ecological outcome, 3) in the case of subdivision or parcelization the parcel identifies a unit of land change, 4) parcel size is the dominant factor affecting land value (Nelson

1988, Throsnes and McMillen 1998), and 5) the parcel has been deemed by some as the "preferred unit for land monitoring" (Mouden and Hubner 2000). I focus the analysis on residential land parcels where properties have been identified as single family residential land use, whereby the property is not connected to urban infrastructure, such as a sewer system.

Contemporary parcel data for Scio and Pittsfield Townships were obtained from the Washtenaw County Metropolitan Planning Commission (2003). I selected out all parcels classified as single family residential from these data. Parcel data for Ray Township (2003) were acquired directly from the Macomb County Planning and Economic Development Department. Similarly, I selected a subset of parcels that represented single family residential land use from these data. Residential land use for Ray Township was determined by aerial photograph interpretation.

To remedy the lack of archived residential parcel data, I digitized the location of built structures from aerial photographs from approximately 1950 to 2000. I then selected all parcels from 2003 that "completely contained" built structures from 1990 to obtain a set of residential parcels that I could be confident did not change in size over time from 1990 - 2003. I repeated this process for each decade to obtain a set of parcels that I could use to record the landscape dynamics within the parcel.

Some inaccuracy in parcel boundary data existed due to the incorporation of setback polygons that separate the parcel into two polygons; one representing the area of the setback as specified by the township and the other representing the core parcel area. Each township had its own rules for inclusion of setbacks in the parcel data. Where setbacks were obvious at the front of the parcel they were excluded. Where they were at the back of the parcel I included them in the parcel calculations because they were useful for adjacency measurements. Additionally, my backcast of 2003 parcel data to the 1950s assumed that residential parcels maintained integrity over time. It is possible that at some earlier time period a residential parcel was part of a much larger parcel that was subdivided to create the one found in 2003. Using photo interpretations and the interpreted structures, I removed any obvious occurrences of this problem and I am confident the associated error from my parcel integrity assumption is low.

As a final discussion point, the stratum of residential properties used in this paper was defined based on parcel size. Residential properties were selected from cadastral data acquired from individual townships who defined individual parcels as residential or some other land use. However, in the above 20 acre category several parcels were found around 50 to 65 acres in size. The classification of such large parcels as residential begs the question at what size can we continue to delineate property as residential? Should these parcels have been classified as farmland or as some other land-use classification? Other research has been conducted to address similar problems relating to the location of exurban sprawl (Ban and Ahlqvist In Press) but not issues at finer scales such as residential property.

2.4 Analysis

I stratified the land-cover analysis by township and parcel size and then present the outcome of all townships collectively. A frequency distribution of the parcel data illustrated natural breaks in parcel sizes at 1 (0.405 ha), 2 (0.81 ha), 3 (1.22 ha), 5 (2.03 ha), 10 (4.05 ha), and 20 (8.1 ha) acres (Figure 4–3). Therefore, I stratified the analysis by parcel size using the following bins (in acres): 1-2, 2-3, 3-4, 4-6, 6-9, 9-11, 11-20, and 20+ whereby the parcel must have been greater than the lower bin value and less than or equal to the upper boundary value. While the definition of exurban development emphasizes the large parcel sizes associated with exurban lots (Daniels 1999, Maestas et al. 2003, Berube et al. 2006), I only excluded those lots less than 1 acre since these are typical of urban/suburban areas and are more likely to have connections to urban infrastructure.

Another determinant of land cover is how the land is used by the land owner. Analysis of the proportion of different land-cover types by land use provides a rough representation of land-owner behaviour with respect to land cover. Because there is variability in land-owner behaviour, I was interested in quantifying not only of the amount, but also the degree of variation and under what conditions variation in land-cover proportions are found to be high or low. I used map algebra (in what is known as

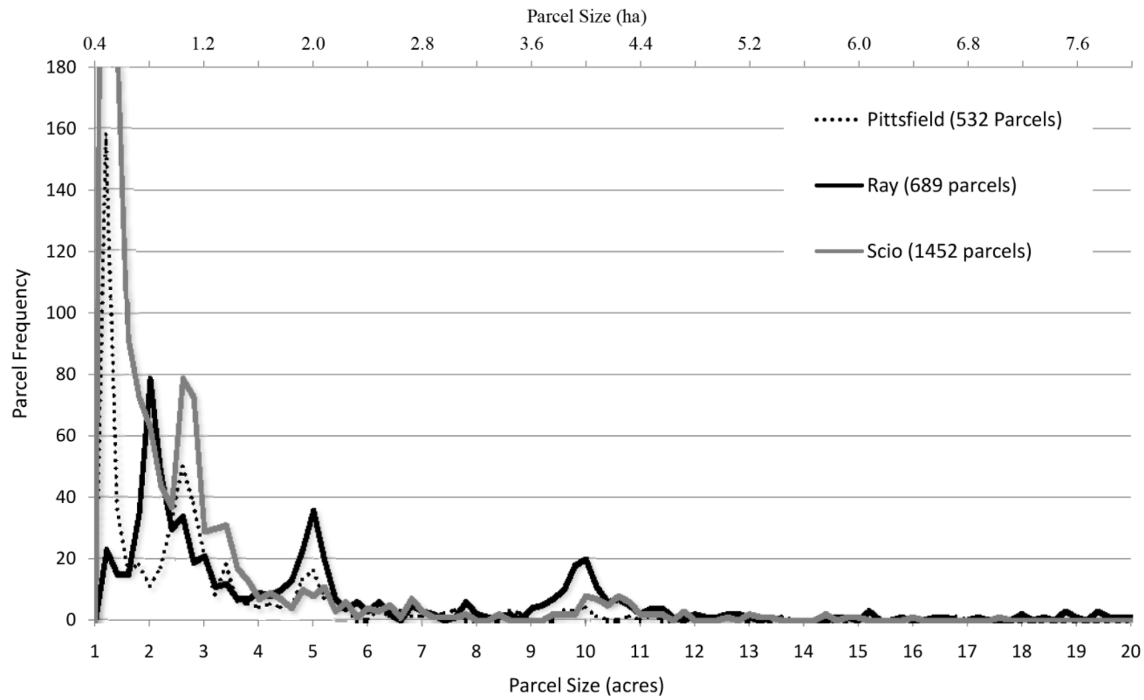

Figure 4–3: Distribution of parcel sizes by township. Mean parcel size for each township is 2.8 acres (1.13 ha) in Pittsfield Township, 6.1 acres (2.47 ha) in Ray Township, and 2.5 acres (1.01 ha) in Scio Township.

a map overlay, Brown and Duh 2004) to identify land cover within each of the sampled residential parcels. I was then able to summarize the quantity and variation of each land-cover type to obtain a distribution of land-cover associated with a single land-use type, residential parcels.

Ecological impacts of land covers are determined to some degree by the configuration and fragmentation of land-cover patches (Li et al. 1993). Therefore, in addition to describing land-cover proportions, I describe the configurations of land-cover types within residential parcels using three fragmentation indices (number of patches, mean patch size, and edge density) and one shape measurement, area weighted mean shape index. Together, number of patches and mean patch size provide a measure of the fragmentation of land-cover classes within residential parcels. A high number of patches and low mean patch size provide evidence of high fragmentation. Conversely a low number of patches and a large mean patch size provide evidence of lower fragmentation.

Edges can affect the functioning of ecosystems (Smithwick et al. 2003, Herbst et al. 2007, Yadav and Malanson 2008, Robinson et al. Accepted – Chapter Three). Edge density, calculated as the total edge of a land-cover type divided by the total area of analysis (i.e. parcel), further facilitates measurement of fragmentation. Parcels with high edge density exhibit more fragmented patterns (within the ownership boundary but not necessarily at the coarser resolution of the subdivision), while those with lower values are less fragmented (Ming-shi et al. 2008). Similarly, the shape of a patch influences the amount of edge and interior patch area. I used the area weighted mean shape index (AWMSI) to describe the mean patch shape within residential parcels. The AWMSI has a value of one when the patch is most compact (i.e. a circle for vector analyses or a square for raster calculations, Rempel et al. 1999, McGarigal et al. 2002). The AWMSI value increases with increasing distortion from these two geometric shapes.

Because land owner behavior and land covers in one parcel may be determined by land covers in neighboring parcels (Nassauer et al. In review), I used local indicators of spatial autocorrelation (LISA) statistics to determine the degree of spatial autocorrelation of land-cover quantity among residential parcels. Specifically, I calculated the local Moran's I for each land-cover type in each parcel to determine if the same land-cover types dominate adjacent parcels (positive spatial autocorrelation, positive value) or if the quantity of land-cover types differ (negative spatial autocorrelation, negative value) among adjacent parcels (Fotheringham 1997). I chose the "Queen's case" for developing the contiguity weights for calculating the Moran's I. In the Queen's case a neighborhood consists of the host polygon and any other residential land-use polygons (greater than 1 acre) that share a point or common boundary.

3. Results

3.1 Distribution of land-cover in residential parcels

The effect of parcel size on the quantity of land-cover differed by land-cover type. Tree cover increased with increasing parcel size (Figure 4–4). The proportion of parcels classified as impervious (i.e. driveway, buildings) decreased with parcel size. Similarly, maintained lawn decreased from 41.0% (on average) of the area of small parcels (1-2 acres) down to 5.4% (on average) for parcels of 20 acres or larger. Perhaps the most interesting trend observed was that of other natural areas. The proportion of other natural areas in small and large parcels (i.e., 15 – 20%) was lower than that in medium range parcel sizes (i.e., 25 – 30%).

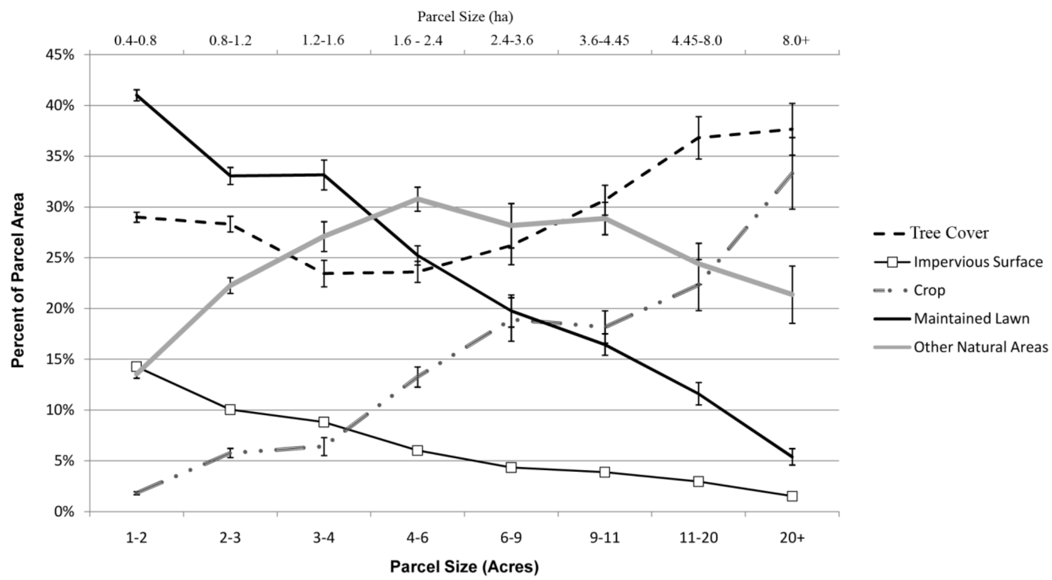

Figure 4–4: Within-parcel mean proportions of land-cover types by parcel size. Error bars represent one standard error and where they are not visible they are very small.

3.1 Land cover patterns

The effect of the number of land-cover patches by parcel size shows that impervious and maintained lawn typically formed fewer than two patches regardless of parcel size (Figure 4–5). Similarly the number of patches in crop rarely increased above one, except for parcels greater than 20 acres. The number of patches classified as other natural area increased slightly with parcel size but typically remained less than three in total. The only land-cover type to experience substantial change in number of patches with increasing parcel size was tree cover. The number of tree cover patches increased from (on average) less than 2 for small parcels (1-2 acres) to around 5 for parcels greater than 20 acres.

While the number of patches did not change with parcel size for all land-cover types, all land-cover types (except impervious) experienced an increase in mean patch size with increasing parcel size (Figure 4–6). The mean patch size of crop (R^2 = 0.96), tree cover (R^2 = 0.98), and other natural area (R^2 = 0.91) increased exponentially with increasing parcel size. Maintained lawn mean patch size increased linearly (R^2 = 0.90) from 1413 m^2 to 4800 m^2.

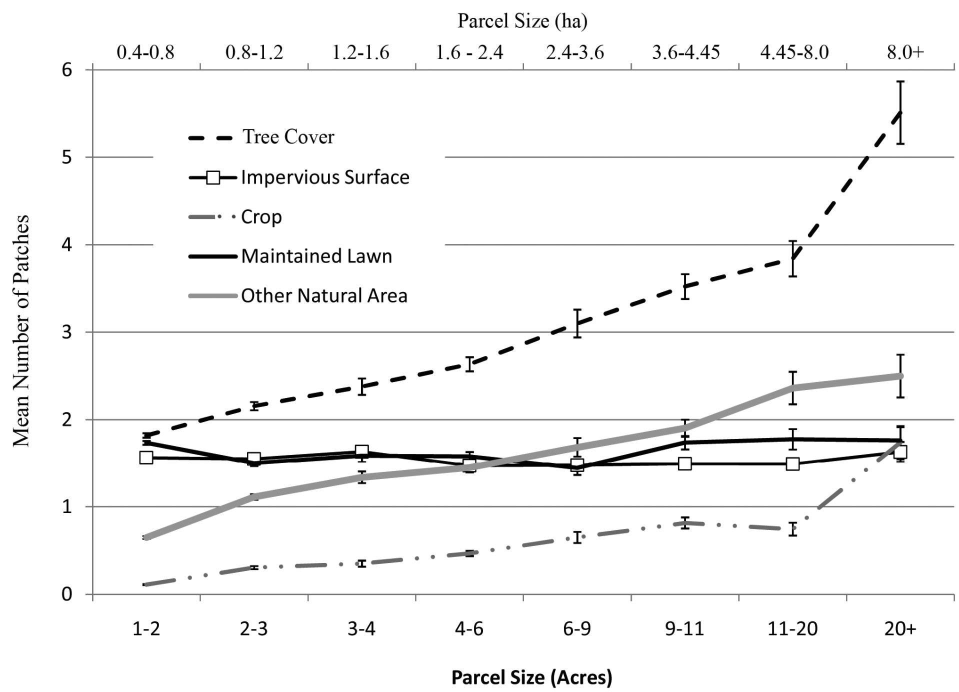

Figure 4–5: Within-parcel mean number of patches of each land-cover type by parcel size. Error bars represent one standard error and where they are not visible they are very small.

Edge density decreased for most land-cover types. Tree cover experienced the highest edge density values, signaling that tree cover was more fragmented than all other land-cover types. While the edge density of tree cover decreased strongly with increasing parcel size up to 4-6 acre parcels, further increase in parcel size had less of an effect. Maintained lawn experienced the largest decrease in edge density with increasing parcel size, suggesting that with larger lots lawn became less fragmented. Similarly, the edge density of impervious surface also decreased with parcel size. Other natural areas initially increased, probably due to an increase in the number of patches of that land-cover type, but then decreased with increasing parcel size. Areas in crop maintained a relatively consistent edge density with increasing parcel size, which is most likely attributed to the consistent errors associated with parcel boundary overlay with neighboring cropland.

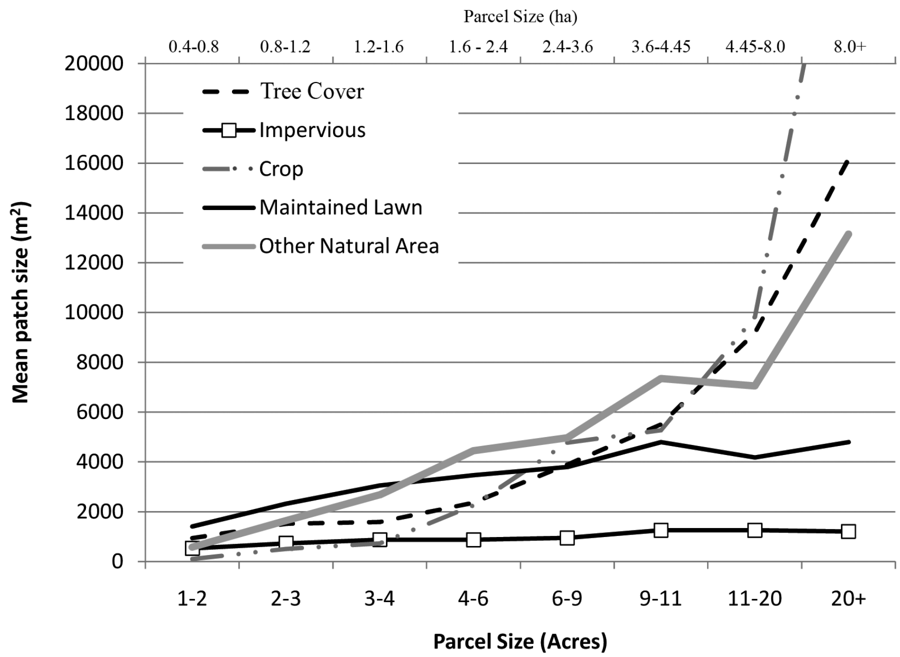

Figure 4–6: Within-parcel mean patch size of land-cover types by parcel size. Error bars show one standard error and where they are not visible they are very small.

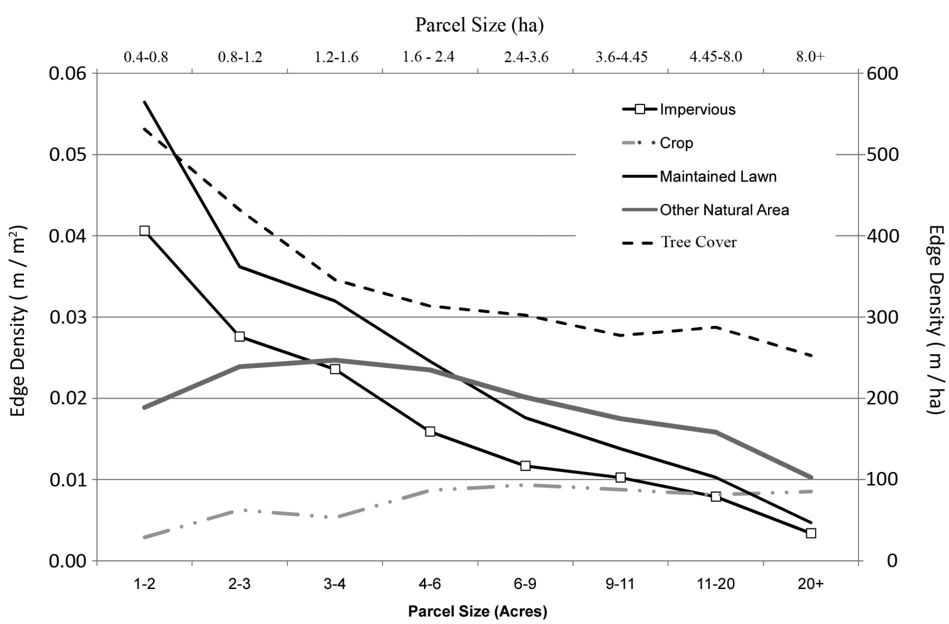

Figure 4–7: Within-parcel edge density of land-cover types by parcel size. Error bars show one standard error and where they are not visible they are very small.

All land-cover types became less compact with increasing parcel size. Other natural areas, which had few patches, held the most compact shape with increasing parcel size. The shape of maintained lawn patches experienced a slight decrease in compact shape, but changed the least of all land cover types. Tree cover experienced the largest deviation from compact shape with increasing parcel size, while impervious surface held the highest deviation from a compact shape at all sizes, which also increased with increasing parcel size. Crop patches increased in compaction and held values less than one due in part to the low amount of area at small parcel sizes.

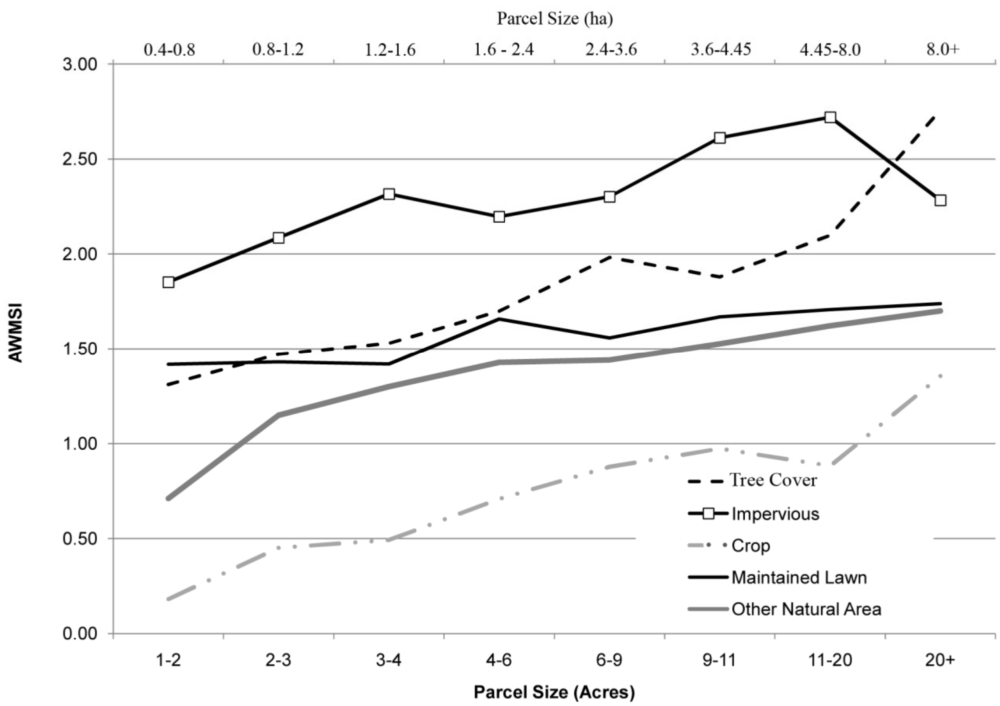

Figure 4–8: Within-parcle area weighted mean shape index of land-cover types by parcel size. Lower values identify more compact patches. Error bars show one standard error and where they are not visible they are very small.

3.2 Spatial autocorrelation of land-cover among parcels

Moran's I values were positive but near zero for both maintained lawn and impervious surface, suggesting that there was only slight spatial autocorrelation between the quantity of each land-cover type and that in neighboring parcels (Figure 4–9).

However, the average Moran's I value for each of tree cover, other natural area, and crop land-cover types increased with increasing parcel size. The most substantial increase was the exponential increase in Moran's I values associated with tree cover at larger parcels ($R^2 = 0.98$). What this translates to on the ground is that larger parcels that have other natural areas, crop, or tree cover on their property are more likely to have neighbors that also have similar quantities of those land covers. Whereas, smaller properties are more likely to have neighbors that have a range of quantities of the same land-cover.

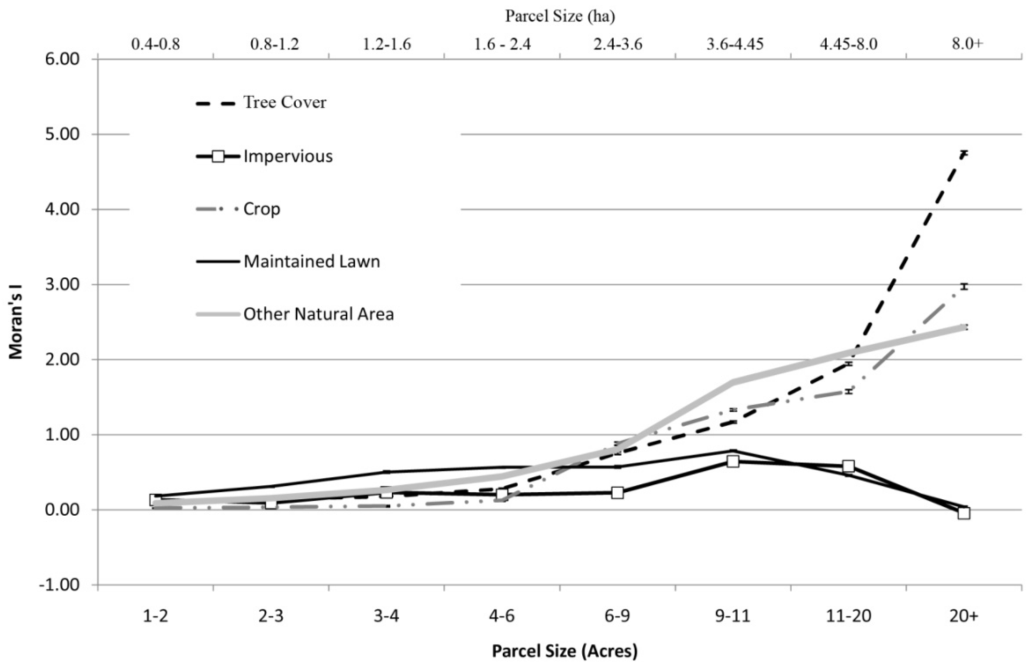

Figure 4–9: Mean Moran's I calculated on land-cover quantities from host parcel to adjacent residential land-use parcels by parcel size. Parcels adjacent to host parcel may be of any size. Error bars show one standard error and where they are not visible they are very small.

4. Discussion and conclusions

The results presented here describe the relationships between parcel sizes and the quantity, fragmentation, and autocorrelation of land-covers in residential parcels located in Southeastern Michigan. With increasing parcel size the proportion of the parcel in tree cover increased and this increase led to more patches, increased patch sizes (on average), and increased tree-cover edge, which collectively created less confined or compact tree-cover patches within residential land-use parcels. Furthermore, large parcels with high amounts of tree cover were more likely located adjacent to other parcels with similar amounts of tree cover. The implications of these patterns are that larger lots typically have proportionately more standing biomass and carbon storage than smaller residential parcels. These results suggest that it may be possible to obtain certain ecological benefits through increased parcel size (e.g. increased carbon storage); however, these benefits need to be evaluated against vehicle miles travelled, energy consumption and air pollution, infrastructure and public service costs, loss of resource lands, impacts on central cities and downtowns, and psychic and social costs (Ewing 1997).

Results for impervious surface differed from the tree cover results. Specifically, impervious surface increased little with increasing lot size so that the proportion of the parcel in impervious decreased, the pattern remained relatively consistent among lot sizes, and the shape of impervious patches were not compact, on average having a larger ratio of edge to area than other land-cover types. Similarly the amount of area in impervious on one property was only slightly autocorrelated to the amount of area on adjacent properties. These patterns suggest that impervious land cover, which typically consists of housing structures and a driveway from the road network to those structures, is located near the front of residential parcels. If impervious land cover was located near the back of parcels then the amount of impervious surface would maintain or increase in proportion to lot size. This suggests that the majority of the natural system is active at the back of the property and land-use policies may be used to further create corridors or contiguous patches of tree cover or habitat at the back of residential properties.

These results provide needed data on the extent of impervious surfaces in residential properties that could be used to evaluate the possible effects of policies aimed at reducing impervious surface and surface-water runoff (e.g. incentives for using porous paving). Further research describing target values for impervious surfaces at varying parcel sizes would provide additional use to policy makers. While research within ecological boundaries (e.g. watershed) denote broadly that declines in ecosystem health occur when 10% of the watershed is in impervious cover and that after 30% is in impervious ecosystem health is severely impaired (Stone Jr 2004), it is not known how these results transfer to the legal property boundaries of residential parcels. If I was to directly follow these guidelines, this research suggests that a focus on reducing impervious surface and surface-water runoff on small parcels is important since they have greater than 10% in impervious. Lastly, a note of caution in interpreting these results, this research describes land cover within residential parcels and not land cover beyond the parcel boundary. Therefore, while the impervious surface results corroborate other research (e.g. Stone Jr. 2004) it does not include road frontage or associated road networks, both of which contribute to the negative ecological effects of exurban impervious surfaces.

The proportions of parcels in maintained lawn decreased substantially as parcel size increased. I expected this to be the case since lawn maintenance can be both capital (e.g. irrigation and fertilizer; UW 2001; Robbins et al. 2001) and labor intensive (Dannenberg et al. 1989; Bittman et al. 2004). The number of patches of lawn was typically less than two and while the mean patch size did increase slightly with increasing parcel size, the amount of edge decreased and patches held relatively the same compact shape. The measurement of the distribution and variance of maintained lawn are particularly relevant since it is estimated that maintained lawn now occupies more land area in the United States than major food crops such as barley, cotton, and rice (Robbins and Birkenholtz 2003). Understanding the patterns of maintained lawn can shed light on the positive and negative ecological effects of lawn in residential areas and with the reported data I can begin to model the ecological effects of lawn using the same tools and in a similar manner as was done in Chapter Three for carbon storage (Milesi et al. 2005).

Given the context set by Theobald (2005), that the rate of exurban growth is 25% greater than population growth, and Ellis and Ramankutty (2008), who demonstrate that anthropogenic biomes cover more than three quarters of the Earth's ice-free land, additional empirical knowledge of anthropogenic landscapes at high resolution is needed. This analysis was conducted at the parcel level because little if any literature or empirical work exists that describes land-cover patterns within exurban residential land parcels. I place this research within the broader context of a larger project designed to explore the ecological effects of land-use and land-cover change. Within this project, called SLUCE, these data are being used to empirically inform distributions of land-covers created by virtual residential agents representing real-world actors in agent-based models of residential settlement and development (e.g. Brown et al. 2008; Robinson and Brown In

Press). Specifically, the data presented here are being used to provide a link between agent behaviors and ecosystem functions.

While the analysis has focused at the parcel level, one must acknowledge that the biophysical world is not always constrained by human systems. Therefore the description of land-cover within groups of parcels, subdivisions, townships, and counties may have slight differences based on the processes operating at each scale and across scales. Interactions among adjacent properties were only briefly touched upon using the Moran's I statistics, however, one could hypothesize increased interaction and feedbacks resulting from aggregate land-cover patterns formed by adjacent properties (e.g. tree-cover corridors created at the back of large adjacent lots). At the micro-level many questions about land-use and land-cover dynamics in exurban areas remain.

Despite that these data are only from Southeastern Michigan, we can infer that some similar patterns will hold in other regions of the United States. While climate, soils, and landform may influence the types of ecosystems residing on the landscape, we can assume that the response of area in impervious surface is similar. I also speculate that while the quantity of maintained lawn may be less in areas like the southwest, I anticipate patterns will be similar with a low number of lawn patches that are compact in shape. The results also suggest that there are limits to the anthropogenic management of land within parcels with increasing size. Therefore, regardless of the ecosystem type (e.g. grassland, savanna, or evergreen tree cover), it is likely that parcels will experience relatively similar quantities of natural vegetation at the back of the property and more managed vegetation elsewhere on the property.

The presented analysis is empirically focused on capturing the heterogeneity and interaction effects of land-cover characteristics among residential parcels so that the findings may also be used to inform contemporary process-based modeling efforts focused on the location decisions and biophysical landscape alterations associated with residential households (e.g. Brown and Robinson 2006, Robinson and Brown In Press). By capturing individual household behaviors and their neighborhood interactions, we can begin to understand how these processes aggregate to create land-cover patterns at the regional scale and how they might be manipulated to mitigate the effects of LUCC on climate.

5. Acknowledgements

Support for the presented research in the form of grants was provided by the Graham Environmental Sustainability Institute (GESI) at the University of Michigan, and the National Science Foundation (BCS-0119804 and GEO-0814542) for Project SLUCE. The author would also like to acknowledge with gratitude the intellectual support of Daniel Brown, Jason Taylor, Lesslie Garrison, Owen Campbell, Daniel Mehdi, and Thomas Simmons.

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Chapter 5

1. Summary

The study of coupled human-environment systems involves research on anthropogenic (e.g. exurban growth) and environmental (e.g. biogeochemical cycling) processes, as well as the interactions and feedbacks between both processes. The overarching goal of this dissertation has been to improve our understanding of the coupled human-environment system that is composed of land-use and land-cover change in Southeastern Michigan with a specific focus on forest cover and carbon storage. The Global Land Project (GLP, Moran et al. 2005) was formed to direct research initiatives toward the study of coupled human-environment systems of this sort. The GLP, which has virtually the same overarching goal as this research, i.e. "to measure, model and understand the coupled human-environmental system," directs and facilitates research initiatives with this goal across the planet. The GLP science and implementation plan establishes three thematic areas to focus research: "(i) the dynamics of land system change; (ii) the consequences of land system change; and (iii) integrated analysis and modeling for land sustainability." Chapters Two and Three of this dissertation directly address the first two of these themes.

By evaluating the effects of different land-use policies on aggregate forest cover in a Southeastern Michigan township, Chapter Two presents an agent-based model that I used to computationally formalize the dynamics of land-system change along the urban-rural fringe (i.e. exurbia). The chapter provides novel contributions to the land-change-science community in the form of a framework for evaluating land-use policy effects on development and land cover, an approach to integrate an agent-based model with a geographic information system (GIS), an example implementation of a United States Department of Agriculture methodology for mapping aesthetic quality, and a number of new examples of methods to empirically inform an agent-based model.

The third chapter of this dissertation evaluated the effects of within-patch and landscape-level heterogeneity in air temperature due to forest fragmentation on carbon storage in Dundee Township, Southeastern Michigan. Effectively this third chapter addresses the consequences of land-system change on carbon storage in a highly fragmented and human-dominated landscape.

New insights from this research showed 1) the inclusion of within-forest-patch air-temperature heterogeneity can significantly influence carbon storage estimates, so much so that when the results of Chapter Three were applied to the conterminous U.S. carbon storage estimates were on the order of current estimates of the North American carbon sink (~1.7 Pg C yr^{-1} in the 1990's, Fan et al., 1998; ~0.65 Pg C yr^{-1} from 2000 – 2005, Peters et al. 2007; ~0.505 Pg, CCSP 2007), 2) that carbon storage estimates increase logarithmically with increasing forest fragmentation when only within-patch air temperature heterogeneity is considered, and 3) the utility of integrating GIS and BIOME-BGC for collection of site data and display of ecosystem model results.

Results from the fourth chapter describe the distribution, fragmentation, and spatial autocorrelation of land-cover types within residential parcels in Southeastern Michigan. While Chapter Four does not address the dynamics of land-system change, it does focus on the dynamics of land cover associated with increasing parcel size in residential land use. Quantification of land-cover characteristics at the residential-parcel level has rarely been conducted. The fine resolution data created in Chapter Four helps to fill this void and remedies limitations found in Chapter Two and Three. Specifically, Chapter Two analyzed township level tree cover resultant from land-use policies that affected subdivision developer behaviors. Results from Chapter Four would allow this to be extended to include policies that affect individual residents and their behavior as it affects land-cover quantity and pattern that subsequently effect ecosystem functions. Similarly this would also allow for the integration of research and tools from Chapter Three to describe the ecosystem dynamics and measurements of ecosystem functions.

Clearly, integrating "(i) the dynamics of land system change; (ii) the consequences of land system change", from above, describes the overall goal of this dissertation and that of the GLP. However, in both Chapters Two and Three the influence of either the natural or human system was represented exogenously. For example, in Chapter Two I constructed and selected the land-use policies that were evaluated based on the aggregate level of tree cover in Scio Township. In no way did changes in tree cover affect the behavior of agents in the model. Similarly, in Chapter Three, I created the landscape fragmentation patterns to identify how that fragmentation and heterogeneity in within-patch and landscape air-temperature measurements affected carbon storage. In no way did these patterns change, as forced by human or environment systems, in response to carbon measurements.

Therefore, the challenge for successful coupled natural-human systems research, similar to that defined by Pielke Sr. (2004), is to integrate resource specific models, models of human decision-making and behavior, and field observations to identify non-linear feedbacks that can be leveraged to mitigate resource use, land conversion, or carbon efflux. Furthermore, these efforts can help identify thresholds beyond which these systems change and different mitigation efforts are necessary. Effectively, what is required is a holistic view of forest ecosystem carbon storage and its relation to a range of human and environmental drivers (Figure 5–1). A better coupling of natural and human systems can link social models driving LUCC "with their ecological ramifications and feedbacks to society" (Riebsame et al. 1994).

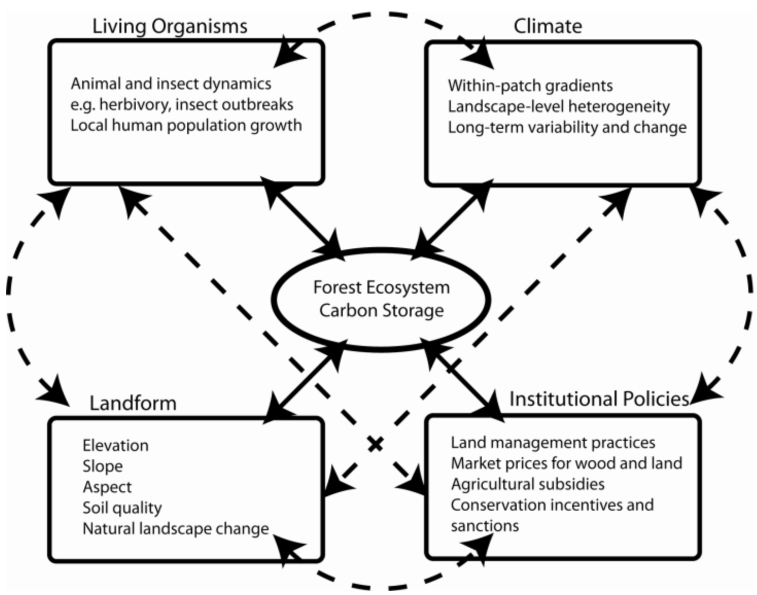

Figure 5–1: Relationship among forest ecosystem carbon storage and human and environmental drivers. Solid lines represent direct forcings on forest ecosystem carbon storage while dashed lines represent indirect forcings.

2. LINKING ECOSYSTEM AND LAND-USE MODELS

Most contemporary LUCC models are concerned primarily with predicting or explaining the quantity and distribution of land cover within a study area (Parker et al. 2003, Verburg 2004). Chapter Two describes the DEED model, which produces typical output measurements provided by LUCC models (e.g. area developed and area in forest cover). A key assumption of the DEED model is that all forest is either full canopy forest or mature trees after 20 years of growth. However, measurements of this type fail to acknowledge the variable quality and function of forest ecosystems and habitat. Fragmentation, overuse, water and temperature stress, loss of biodiversity, noxious/invasive weed infestations, and noise, air, and water pollution are some of the anthropogenic factors that can degrade the quality and efficiency of ecosystem functions. None of these factors are accounted for in Chapter Two.

In contrast, Chapter Three specifically addresses one ecosystem function (i.e. carbon storage) and how it varies with air-temperature change from the forest patch edge to the interior and with variable air temperature and fragmentation of forest cover across the landscape. The chapter not only evaluates these effects on carbon storage but also provides an implementation of a "big-leaf" ecosystem model (i.e. BIOME-BGC) in a way that it may be used in combination with a LUCC model. Comparing the outcome of the two chapters demonstrates a trade-off between the estimation of area in forest on the one hand and the estimation of forest quality or function on the other.

The coupling of a LUCC model, like DEED, with a forest ecosystem model, like BIOME-BGC, requires the integration of processes operating and linked across a variety of spatial and temporal scales (Figure 5–2). Linked processes operating at different temporal and spatial scales do not produce smooth linear state transitions. At best we can imagine that ecosystem functions (e.g. carbon storage) decrease at a decreasing rate with increasing human impact on the land. At first the effects of incremental development on carbon storage are large and negative (e.g. land clearing and burning of forest cover), but they then stabilize over time with little further influence on carbon (Figure 5-3a). However, it is also possible that development has an increasingly negative effect on carbon storage (e.g. land clearing for subdivision spawns neighboring areas to develop) until at some point the functioning of an ecosystem cannot be easily degraded. Perhaps at this point the forest shifts entirely to maintenance functions that are more resilient to stress, as opposed to new growth, which is more susceptible. Alternatively, perhaps the amount of area for development has neared exhaustion (Figure 5–3b).

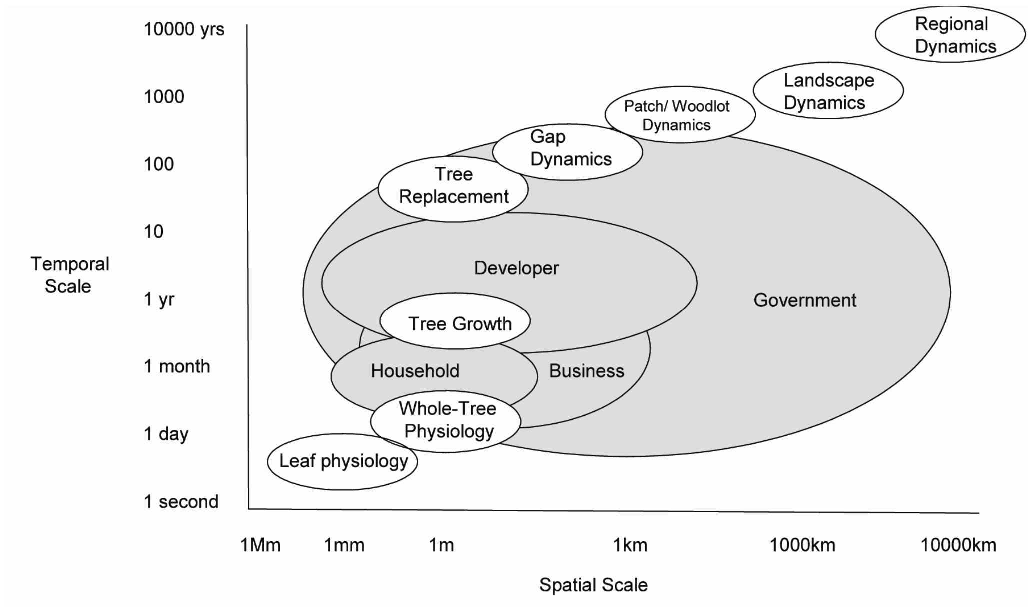

Figure 5–2: Nested hierarchical representation of carbon/biomass dynamics in a forested landscape from King (1991), adapted to include the temporal and spatial scale of actors driving land-use and land-cover change. White ellipses denote original biophysical processes from King (1991), while, grey ellipses denote anthropogenic actors of land-use and land-cover change.

Some researchers have argued that it is most likely that an ecosystem is resilient to some degree of influence from land development whereby the ‘health’ of the ecosystem takes a sudden and dramatic shift when passing a critical threshold (Eiswerth and Haney 2001). Known as punctuated equilibrium, these shifts could cause major changes in ecosystem species composition (He et al. 1999) or susceptibility to disturbance. While these points are generally theoretical in nature, only by linking LUCC and ecosystem models can we begin to explore these types of questions and better understand the anthropogenic effects on biophysical properties and functions of ecosystems.

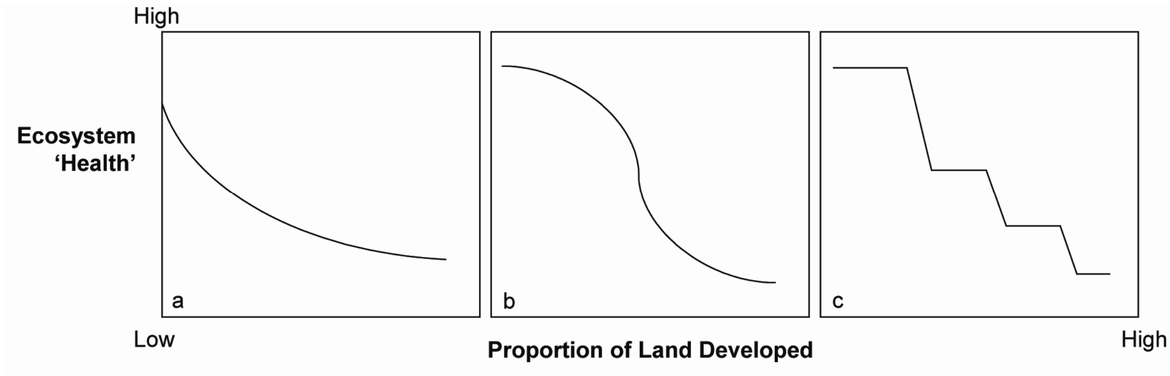

Figure 5–3: Possible trends of ecosystem 'health' as related to amount of development within proximity. Adapted from Power et al. (2005), a) shows greater initial effects of development on ecosystem 'health', b) shows increasing ecosystem degradation with increasing development until some point of inflexion whereby increasing development has a degrading effect but at a decreasing rate on ecosystem health, and c) characteristic of punctuated equilibrium whereby the ecosystem may take a sudden shift to a new state of highly degraded health as a result of increasing development.

3. FUTURE RESEARCH

The next steps of the research program begun in this dissertation involve linking the components of Chapters Two, Three, and Four, such that the LUCC model creates land-cover patterns at the parcel scale and subsequently produces quantitative measurements of ecological function at the landscape scale. Effectively, I would then be able to capture both the quantity of area in forest and the quality of that forest as it pertains to different ecosystem functions, as described above.

Technical developments. Yadav et al. (2008) describe three different methods of integrating biophysical models with agent-based land-change models: restricted choice coupling, semi-integrated coupling, and fully integrated coupling. The restricted choice method involves predefining, with the ecosystem model, all possible biophysical outcomes associated with all possible LUCC outcomes (i.e. management and site characteristic) and placing these results in a look-up table. Then for a given land use and land cover, the model can look up the associated biophysical outcome. The semi-integrated coupling involves having the LUCC model interrogate landscape data (i.e. site characteristics such as soil texture) that are held constant throughout the model runs and rerun the ecosystem model when LUCC changes occur. In a fully integrated coupling the landscape data may change, within a model run, requiring the LUCC model to store these changes and use the new landscape data in subsequent model steps.

In Chapter Three I loosely coupled a GIS and BIOME-BGC, preprocessing data in the GIS for input to BIOME-BGC and then displaying BIOME-BGC output back in the GIS. Tightly integrating BIOME-BGC within ArcGIS would permit specific feature areas (i.e. points, polygons, or sets thereof), with unique site characteristics, to be run with BIOME-BGC to observe results geographically. A framework of this type sets the stage for a semi-integrated coupling approach (Yadav et al. 2008) with the DEED model presented in Chapter Two. To create a fully integrated system, as described by Yadav et al. (2008), would require the storage and alteration of parameters and nutrient pool values while the land-use and land-cover change model was running as well as a step-wise progress update of BIOME-BGC that coincided with the steps of the LUCC model.

Conducting these technical advancements in coupling natural and human systems can enable us to address a host of new questions such as the following: at what spatial and temporal resolutions do ecosystem processes coincide with human decision making? What ecological drivers and changes in ecology directly influence human behavior? Through what avenues or pathways do human and natural systems influence each other strongest (i.e. through visual, then economic, then moral)? What types of actors or decision makers most affect ecosystem function and what ecosystem functions affect what actors?

Integrated research. A fully coupled natural-human system as described above could assist the further development or evaluation of existing theories, like those posed in Figure 5–3. By integrating geographic, ecological, sociological, economic, and regional science concepts and theories we can begin to generalize the effects of development on ecosystem functions at local and landscape scales. Then we can begin to understand what conditions lead to a decline in specific ecosystem functions and how policy may be used to mitigate these declines. Furthermore, if we integrate collective action research into this framework we may be able to model the degree of community involvement among residential land-owners that may subsequently influence community development, i.e. like public land purchases that could produce positive feedbacks in land use development. For example, early residents that secure large lot parcels could vote to influence park creation in their neighborhood that results in high environmental quality in one region, but which may create areas of low environmental quality in another. I would expect this type of result to occur when information diffusion exists among residents that could coordinate a critical mass for environmental protection and public land purchasing occurs in a strictly residential area, versus a residential neighborhood that is fragmented by other land-uses (e.g. commercial/industrial). These cross scale interactions between residents and government may feedback to cause significant differences in the configuration of land use and land covers as well as the amount and distribution of carbon storage.

Application. Ideally, the integrated framework would be used to assist real-world decision makers. The modeling of human and environment processes that incorporate realism within a GIS provide a framework for consultation and understanding with stakeholders and interested parties. The ability to work with NGO's, developers, home-owners and stakeholders, and policy makers within a framework that utilizes site-specific maps and data to which they are familiar provides a foundation for discussion and collaboration so that products such as those presented in this dissertation may be used beyond the walls of academia. For example, the products of this dissertation may facilitate the identification of possible sites for conservation based on desired ecosystem functions such as maximizing carbon storage. Substantial research on conservation location strategies exists (e.g. Prendergast et al. 1999, Wang and Medley 2004) and an integrated ecosystem model and LUCC approach could improve the ecological awareness and credibility of the choices and policies implemented by land-use decision makers. It is my hope to further the research presented in these pages such that I can take the knowledge and tools gained within and aid the development process to suit the needs of the affected communities.

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