LOCAL GROWTH AND LAND USE INTENSIFICATION: A SOCIOLOGICAL STUDY OF URBANIZATION AND ENVIRONMENTAL CHANGE

by

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DISSERTATION ABSTRACT

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This dissertation takes a sociological look at the relationship between urbanization and environmental change. While sociological studies on urbanization have long addressed the social dimensions of the built environment, the natural environment has not been treated as a primary concept in urban sociology. Based on an analysis of local land use change across the United States at the beginning of the 21st century, this dissertation brings the built and natural environments together, recognizing both as important dimensions of urbanization. The expansion of the built environment, through deforestation and the covering up of fertile agricultural land, represents a modern form of land use change with direct and indirect impacts on the natural environment, the most severe effects of which are seen in biodiversity loss, disruption of the nitrogen cycle, and climate change.

Drawing on literatures and theories in environmental, rural, and urban sociology as well as demography and human ecology, the bulk of the dissertation involves empirical analyses of overall changes in forest cover as well as the loss of forest cover and agricultural land to the built environment (i.e., the impervious structures and surfaces that cover the land), a process I refer to as land use intensification. My dissertation project uses quantitative methods to examine the demographic, economic, and social

forces behind this process in contemporary America. Hypotheses are derived from the various literatures mentioned above; to test these hypotheses, I integrate county-level data from US governmental sources with satellite imagery on land cover change from the National Land Cover Database (NLCD). For the years 2001-2006, I use the NLCD data to quantify three dependent variables at the county-level: overall change in the area of forest cover as well as the area of forest cover and agricultural land lost to the built environment. Results from regression analyses demonstrate that urbanization is a multidimensional process that differentially transforms the American landscape. With a focus on land use intensification, this study advances a sociological framework to address connections between urbanization and changes in both the built and natural environments.

This dissertation contains previously published and unpublished co-authored material.

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CHAPTER I

INTRODUCTION:

URBANIZATION, LAND USE, AND ENVIRONMENTAL CHANGE

Human life swings between two poles: movement and settlement - Lewis Mumford (1961)

The modern [age] is the urbanization of the countryside, not ruralization of the city as in antiquity - Karl Marx (1857)

The transition to capitalism alienated the products of the land as much as the products of human labor, and so transformed natural communities as profoundly as it did human ones - William Cronon (1983)

Excessive private consumption was not inevitable. It was the result of sustained pressure from real estate interests and their allies in government to marginalize the alternatives to unlimited private suburban growth. - Dolores Hayden (2003)

Nearly 7,000 years ago humans began to settle densely in constructed spaces that archaeologists and historians today identify as the first cities (Lawler 2012). For most of the time since then, however, the proportion of the world's population living in cities had remained small, fluctuating at most between three and six percent, with higher rates likely during the ancient Arab, Chinese, and Roman civilizations (Grauman 1977; Ponting 2007). Only in the past 150 years or so has the urban population experienced exponential growth (Davis 1955; Satterthwaite 2007); today one-half of the world's population lives in urban areas, and that figure is expected to increase to just over two-thirds by 2050 (United Nations 2014). Moreover, individual cities today are several orders of magnitude bigger than previous urban settlements. Probably only a couple of pre-industrial cities (e.g., K'ai-feng in China and Rome) had populations of one million or more (Chew 2001; Ponting 2007). By 1970, however, there were two megacities in the world, each of which had more than ten million people. As of 2014, the world had 28 megacities; the United Nations predicts the frequency and size of these massive urban centers to grow, with no expectation that the overall pace of urbanization will slow down anytime soon (Biello 2012).

What this brief history suggests is that urbanization and city life are unique and defining characteristics of modern human society. In this dissertation project, I utilize sociology to address the environmental consequences of our urban world. In the bulk of the analysis (Chapters II-IV), I employ quantitative methods to examine the local demographic, economic, and social drivers of land use intensification, a concept used to represent the progressive exploitation of land by modern human activity (Logan and Molotch 2007; Molotch 1976).[1] In particular, I look at the types of environments that are lost in the process of land use intensification, focusing explicitly on agricultural land and forest cover at the start of the 21st century in the United States. In this country, agricultural land and forest cover are two types of landscapes that continue to be

[1] Chapters II and III are revised versions of two previously published articles, and Chapter IV is a revised version of an article currently under review. These three articles were written with co-authors.

transformed by human activity. While the total area devoted to agriculture in the U.S. has continued to decline (USDA 2009), the overall area of forest cover has not changed considerably (Alvarez 2007). Nonetheless, a measure of net change in forest cover obscures not only substantial local and regional variation but also differences with respect to the specific drivers of deforestation. For instance, agriculture is no longer the primary cause of deforestation in the United States, as it once was (Cronon 1983). Rather, in the past several decades, forests in the U.S. are increasingly being cut down for the purpose of urban land development (Foster 2010). To study these land use changes and their demographic, economic, and social drivers, I utilize county-level data covering the years 2001-2006 for both the Southern United States (Chapter II) and the entire continental United States (Chapters III-IV). This wide geographic coverage lets me analyze the systemic sociological processes that contribute to landscape transformation at the local level in a modern, industrialized society. As such, this project yields fresh insight into the socio-ecological dimensions and consequences of our modern urban world, a subject of increasing concern in both the social and natural sciences (e.g., Clement 2010; Ergas 2010; Elliott and Frickel 2013 and In Press; Grimm et al. 2008; Marcotullio et al. 2014; McKinney 2013; Wachsmuth 2012; Worldwatch Institute 2007; Young 2009).[2]

[2] Here, I describe some terminology used in the dissertation. First, following the typology of Dunlap and Catton (1983), the term "built environment" refers to human constructed impervious surfaces (e.g., roads, homes, commercial centers, and industrial facilities), and the term "natural environment" refers to both "undeveloped land"(i.e., land that has not been modified ostensibly by human activity, such as forests and grasslands but excluding areas devoted to agriculture and tree plantations) as well as "biogeochemical cycles" (i.e., the pathway by which an element or molecule moves through living and non-living material, ultimately supporting life on Earth). Next, the term "land use intensification", defined in the above paragraph, is drawn from Logan and Molotch's growth machine theory. In the dissertation, I use it interchangeably with "land development". Meanwhile, "land use change", "land cover change", and "landscape transformation" are more inclusive terms to describe the many ways that humans have modified land throughout history (Ellis 2013; Rudel 2009). Lastly, I use the abstract term "growth" to represent alternately increases in the size of the population, the level of economic activity, and expansion of the built environment. In the conclusion, I elaborate on some concerns with the term "growth", briefly discussing the different ways that urban social scientists have described the political-economic context of modern society,

The main argument of the dissertation is that urbanization is a multidimensional driver of environmental change; the different dimensions of which have countervailing impacts on land use intensification. The bulk of the project (Chapters II-IV) employs quantitative methods (e.g., regression models) to evaluate sociological theories about these countervailing impacts. The different chapters focus on the separate demographic, economic, and social forces behind landscape transformation. Chapter II looks at how rural and urban population change drive changes in forest cover in opposite directions, whereas Chapters III and IV examine the way growth and urbanization, as multidimensional processes, differentially contribute to the loss of agricultural land and forest cover, respectively, to the built environment. Drawn from a variety of sub-disciplines (i.e., environmental, rural, and urban sociology as well as demography and human ecology), several theories are featured in the analyses, including growth machine (Logan and Molotch 2007), metabolic rift (Foster 1999), treadmill of production (Schnaiberg 1980), and ecological modernization (e.g., Ehrhardt-Martinez 1998). Each chapter in greater detail discusses the relevant theories and describes the data and methods used to test hypotheses derived from these theories.

In the remainder of the introductory chapter, I review the relevant socio-ecological literature on urbanization, with a specific emphasis on landscape transformation, which sociologists regard as a basic dimension of urbanization (Logan and Molotch 2007). This review helps to contextualize the geographic focus of the subsequent chapters, framing the environmental changes happening at the local level across the United States within a broader discussion about the socio-ecological

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either using the abstract term of "growth" (e.g., Logan and Molotch 2007) or the more specific term of "capitalism" (e.g., Harvey 1982).

dimensions of urbanization around the world and their global consequences. Moreover, this review highlights the common thread linking all the chapters together, helping to shape the dissertation into a comprehensive analysis concerning the environmental impacts of urbanization. In the remainder of the introduction, I first discuss those impacts in general terms, paying attention to the way urbanization broadly implicates changes in both the natural and built environments in terms of landscape transformation, the use of natural resources, and the concentration of anthropogenic wastes. Second, I then draw on the concept of planetary boundaries (Rockström, et al. 2009) to offer three specific examples of urbanization's ecological effects, looking at biodiversity, the nitrogen cycle, and the carbon cycle. With the planetary boundaries concept, I frame urbanization as a socio-ecological process characterized by both proximate and remote, as well as direct and indirect, environmental changes, all of which are scaling up to the global level, creating problems for the entire biosphere (Grimm et al. 2008). Lastly, to wrap up the introduction, I briefly discuss prospects for ameliorating these ecological impacts, highlighting the argument, from the environmental social sciences, that solutions to these urban problems (e.g., higher residential densities and green technologies) are constrained by the unequal, pro-growth forces of modern society (e.g., Agyeman et al. 2003; Foster et al. 2010; Logan and Molotch 2007; Pinderhughes 2004; Schnaiberg 1980).

The Environmental Consequences of Urbanization

In ecological terms, compared to pre-industrial societies, modern urbanization and the cities we inhabit today present very different problems for humanity (Chew 2001). Certainly, the construction of cities and the demands of city life have always entailed environmental impacts in the form of landscape transformation, natural resource utilization, and the concentration of anthropogenic wastes. Nevertheless, with the rise of industrial capitalism, the scale of these impacts has changed in the past 150 years or so (Foster 1994; Ponting 2007). For instance, while the smelting of lead and copper by the Romans generated air pollution that can be detected as far away as the Arctic (Hong et al. 1994 and 1996), the spread of cities around the world since the Industrial Revolution has resulted not only in local and regional environmental impacts but also impacts to the global biosphere, especially in terms of biodiversity loss as well as the disruption of the nitrogen and carbon cycles (Chameides et al. 1994; Grimm et al. 2008). Again, I elaborate further on these specific issues in the next section; here, I concentrate on how modern urbanization entails general changes in the natural and built environments, and I present landscape transformation as a fundamental dimension in this socio-ecological process of change with explicit and implicit connections to natural resource use and pollution.

Many environmental scholars argue that our modern urban world poses a set of distinct and important questions for the social sciences (e.g., Davis 1998). Nevertheless, all urban settlements throughout history have dealt with issues of landscape transformation, natural resource utilization, and the concentration of anthropogenic wastes. And, to a large extent, these three consequences, and the human activities that generate them, are interrelated in a complex socio-ecological system with feedbacks and reciprocal influences. The transformation of landscapes entails changes in the use of natural resources and the production of pollutants; both of which feedback into and shape the different ways that humans transform the land. One can trace the basic contours of this reciprocal chain of influence with landscape transformation playing a more pressing and prominent role. For instance, when the landscape is transformed to build houses and pave roads, urban dwellers must exploit natural resources that are not immediately available in the vicinity of the growing or emerging city, such as timber and stone (Cronon 1992). Deforestation and the quarrying of stone, for instance, make it possible to the build the physical infrastructure of cities; as this built environment grows, it permits more in-migration and facilitates the dense settlement of humans on a smaller area of land. As humans concentrate on smaller areas of land, so too do the wastes they create, both in terms of humanure and all the other unused by-products from the various human activities that require natural resources (DeFries 2014).

Yet, of these three processes of general ecological change, landscape transformation stands out as a plainly visible consequence of urbanization. Certainly, human wastes and resource use can leave their marks on the natural environment, but the built environment of cities throughout history has left a lasting imprint on landscapes around the world; archaeological digs and, more recently, satellite technologies have helped to detect this imprint in places once thought unsuitable for dense human settlement, such as the Amazon delta (Drake 2012; Mann 2005). Still, the imprints of today's cities are far more noticeable and will persist much longer in the Earth's geological record. In fact, the built environments of cities today, due to urbanization's global reach and the use of synthetic products in construction materials, will form a distinguishable stratigraphic layer in the planet's land system (Zalasiewicz et al. 2011). Indeed, the impending stratigraphy of modern urban society is one reason for distinguishing a separate geological epoch in Earth's history, which scientists have called the Anthropocene, or the "Age of Man" (Vince 2011).

Whatever the period in human history, and whatever the degree of civilization, landscape transformation and especially the construction of the built environment have impacts that are simultaneously direct and indirect as well as short-term and long-term. For example, with the arrival of European settlers and the collapse of indigenous societies, historical and archaeological research on land use change has documented a period of massive reforestation in the Americas; because these re-grown forests sequestered so much carbon dioxide from the atmosphere, climate scientists argue that this reforestation contributed to the dip in global temperatures known as the Little Ice Age (Mann 2011). Yet, over the next century or so, the European settlers in America increasingly cut down these re-grown forests, partly in response to the growing demand for food, fiber, and construction materials from an urbanizing Europe (Bagchi 2005; Williams 2010). Then, with the rise of industrial capitalism in America, the country's cities grew and moved westward, which not only shifted the urban reach for natural resources but also coincided with colossal land transformation and infrastructure projects required to make city-life possible, especially in otherwise inhospitable environments like Chicago and New Orleans (Colten 2006; Cronon 1992).

Throughout this urban transformation, the problem of human waste and pollution in cities was also intensifying. For instance, contemporary social and natural scientists in the 19th century were paying attention to the disruptive effects of unprecedented rates of urbanization on the soil-nutrient cycle (Foster 1999). Even though large-scale urban civilizations of the past, especially in places like Rome, could not adequately address the human waste problem of their cities, many early urban settlements made attempts to do so, understanding the importance of applying urban organic wastes to neighboring farmlands (DeFries 2014). In fact, many pre-industrial cities around the world were *relatively* successful in implementing ecological practices to close the metabolic cycle between society and the soil (Harvey 1996: 411; Mumford 1961: 257). Yet, with the Industrial Revolution, populations shifted to urban centers, and the traditional forms of recycling urban waste back to agriculture largely disappeared. Karl Marx was a contemporary observer of this process and argued that the disruption, or "rift", in the society-soil metabolism was being propelled forward by capitalist forces of private property and accumulation, especially as workers were forced to move to urban centers and sell their labor in the cities' factories (Foster 1999). His critique of capitalism's "metabolic rift" explicitly addressed the connection between declining soil fertility in the countryside and the concentration of human waste and pollution in the city. Marx's analysis of urbanization's environmental consequences was one of the first by a social scientist, and it was framed within the context of an unequal society oriented around growth. Later on in the introduction, I elaborate on how the unequal, pro-growth forces of modern society complicate solutions to urbanization's environmental impacts. Here I briefly note how these forces broadly influence the way cities are built, especially in the United States (Logan and Molotch 2007), emphasizing the implications for landscape transformation as well as natural resource use and pollution.

In the pursuit of growth, the real estate and financial sectors in the United States have long profited from new land development (Baran and Sweezy 1968; Hayden 2003), which, as I outlined above, precipitates a chain of environmental consequences. Indeed, with each newly developed parcel of land (through the construction of houses, roads, commercial centers, and industrial facilities, for instance), society commits itself to a higher level of natural resource use. The increasing use of natural resources is required not only for the maintenance and repair of the new built environment itself but also to fuel all the additional anthropogenic activities happening on this freshly developed land. Moreover, as previously noted, the relationship between changes in the built and natural environments is reciprocal. In order to continue building these impervious surfaces and structures, society needs greater access to a variety of natural resources, like timber and stone, mentioned above, but also increasingly petroleum. In this way, in a pro-growth society, the reciprocal relationship between land development and natural resource use and pollution is akin to what some environmental scholars have called the socio-ecological treadmill (O'Connor 1988; Schnaiberg 1980). Based on the treadmill argument, modern society, in order for it to experience uninterrupted growth, relies on increasing access to natural resources; yet, as society grows, the level of human activity escalates, which then ramps up resource use even more. In the United States, this treadmill-like relationship is perhaps most evident in the connection between the built environment and the widespread use of fossil fuel following World War II. In short, the rise of the automobile both influenced and was influenced by the car-centric urban form that characterizes American cities today (Gonzalez 2009; Sweezy 1973). New cities were designed and old cities redesigned to accommodate the automobile and the lifestyles that became dependent on it (Kay 1997). In this car-city arrangement, the daily rounds of individuals to work, home, and the store could not be done without the combustion of gasoline as well as the continued construction and maintenance of the infrastructure that supports automobile traffic, both within and between cities. This massive physical infrastructure has become a fixed feature in the American landscape, generating higher levels of human activity and natural resource use as a result (cf. Perelman 2006). In order to maintain the viability of this infrastructure, and to continue constructing it, even more resource use is required (Güneralp and Seto 2012). In this light, with the increasing presence of the built environment, the ecological impacts of urbanization and city-life became even more entrenched and permanent. The enduring consequence of land use intensification ultimately complicates any efforts to ameliorate the impact of cities. I elaborate on this general complication below; here, I note that, for instance, when the forces of growth start to slow down (e.g., during a recession), there is not a proportionate reduction in the use of fossil fuel by modern society (York 2012a); the massive physical infrastructure of urbanization today has raised the floor on the minimum level of natural resource use required for the regular operations of our pro-growth society.

At this point, one can see that the environmental consequences of urbanization and city-life include more than just resource use and the pollution that is immediately generated as a result. Indeed, I locate land use intensification at the center of urbanization's socio-ecological complex, acting as a catalyst in a dynamic, reciprocal process of environmental change, the dimensions of which are multifaceted. Historically, many big cities were constructed just above the mouths of rivers, where nutrient rich sediments and the energy from the flow of water created favorable conditions for pre-industrial human settlement (Odum and Odum 2001). Until the rise of fossil fuel use, these ecological constraints continued to play a major role in the location of cities and limited the spread of urban development (Mayumi 1991; Mumford 1961; Odum and

Odum 2001; Smil 1994). Today, urban development continues to take place on biologically diverse lands and productive soils, but these environments are increasingly spared for other uses, either in the form deforestation, as forests are cleared to make space for new impervious surfaces, or through the way the built environment is covering up agricultural land that could otherwise be used to produce food and fiber for humans (DeFries et al. 2010; Imhoff et al. 2004; Nizeyimana et al. 2001). Thus, the permanence of the modern built environment forecloses on the possibility of other types of land use that are environmentally more benign, like undeveloped land and open-space or even agriculture. Additionally, when land is covered up with pavement and buildings, it destroys habitats and has a long-term impact on the availability of carbon biomass, the basic fuel for all heterotrophic organisms that rely on photosynthesis for the production of food, not just humans (Imhoff et al. 2004). Indeed, biodiversity loss as well as the disruption of the carbon and nitrogen cycles are three specific changes that characterize our urban impact on the global biosphere today. On that note, I now turn to a discussion of the planetary boundary concept in order to elaborate on more specific urban environmental impacts and how they are scaling up to the planetary level.

Three Planetary Boundaries

Research by natural scientists indicates that anthropogenic environmental change is undermining the planet's capacity to maintain a safe operating space for humanity (Rockström, et al. 2009). Through a series of related processes – including agriculture, deforestation, and land use intensification – urbanization directly and indirectly contributes to modern society's impact on the planet. As discussed above, cities throughout history have always had various ecological consequences; nevertheless, these consequences have become increasingly global in scale, thereby presenting a challenge to the environmental sustainability of modern society (Barnosky et al. 2012; Foley et al 2005; Rockström, et al. 2009). To underscore the importance of this challenge, I will now discuss the ways in which urbanization is contributing to three particular measures of global environmental change; these are biodiversity loss as well as the disruption of the nitrogen and carbon cycles. While other measures of ecological change are also deteriorating globally, scientists are concerned that the extent and magnitude of the changes to biodiversity, the nitrogen cycle, and the climate might eventually destabilize the biosphere and potentially undermine the environmental conditions in which human civilization has thrived during the past several millennia. To be clear, the health and survival of the planet's non-human species is already being undermined. In this sense, some scholars have suggested that the research behind the planetary boundaries concept is anthropocentric (Moore 2011); but, if this concept is anthropocentric, it is only minimally so because this research suggests that anthropogenic forces are pushing Earth's major life support systems beyond critical thresholds, which pose a threat to *both* human and non-human species. Here, I briefly review three of these boundaries to demonstrate how urbanization, as a multidimensional anthropogenic force, is contributing to planetary change.

Biodiversity Loss

The abundance and diversity of species play an essential role in regulating Earth's life support systems. Nevertheless, the global rate of biodiversity loss today far exceeds the rate in pre-industrial times (Rockström, et al. 2009). In fact, the pace and magnitude of species extinction stand out in the geological record, marking biodiversity loss during the Anthropocene as a unique and defining feature in Earth's history, equivalent only to previous mass extinction events caused by global climate change, plate tectonics, and collisions with extraterrestrial bodies. While difficult to identify precisely, the biospheric boundary in biodiversity loss has likely already been crossed, potentially jeopardizing the broad resource base on which human civilization has depended. Yet, human activities, rather than geological or cosmological events, are driving the rapid and widespread destruction of species today. While urbanization and city-life contribute to this process in multiple and subtle ways, I briefly review the impacts on biodiversity loss caused by land use intensification for the purposes of agriculture and the built environment. Broadly speaking, current patterns of urban development and agricultural practices are transforming Earth's land on an unprecedented scale, destroying species and their habitats in the process.

Globally, the built environment of cities covers a small proportion of Earth's surface, with estimates ranging from under 0.5 percent to nearly three percent of the planet's land area (Hooke and Martín-Duque 2012; Lambin and Meyfroidt 2011; Seto et al. 2012; Sutton et al. 2009). Although this area is relatively small, the residents living in these urbanized spaces tend to rely on the products from industrial agriculture (DeFries et al. 2010). Cropland and pastures cover about ten percent and twenty five percent of the Earth's land area, respectively. Moreover, conservationists note that the ecosystems adjacent to cities tend to contain significant concentrations of species that are native to the region; the expansion of urbanized land threatens the survival of these endemic species (McDonald et al. 2008; Seto et al. 2012). As humans transform land into urban spaces, there is a direct and indirect loss of habitat and carbon biomass, which together contribute to an overall decline in the biodiversity of regions both adjacent to and spatially removed from (but remotely connected to) the world's cities.

Below, in a subsequent section, I discuss the issue of carbon biomass as part of a separate planetary boundary in terms of the carbon cycle; here, I briefly outline urbanization's specific impact on habitats, which is perhaps the most immediately visible environmental change to occur. This change is observed not only as undeveloped land is transformed by industrial agriculture and forestry to clothe, feed, house the residents living in cities (DeFries et al. 2010) but also as the area of the built environment of cities grows. Through the use of earth moving machinery to bulldoze and level the land, and then the covering up of this land with impervious surfaces and structures, urbanization contributes to both the destruction and fragmentation of habitats (Ewing et al. 2005). Cutting down trees, mowing fields, and filling in wetlands are some of the ways in which landscapes are initially transformed, making the local environment suitable for the eventual construction of houses, roads, commercial centers, industrial facilities, and the other impervious surfaces of urban areas. In these ways, the expansion of the built environment reduces the habitable area available for non-human species. Not only do endemic species tend to occupy the areas being converted into urbanized spaces, as noted above, but also, these areas disproportionately tend to be the habitats of the plants and animals that are threatened and endangered. For instance, Ewing et al. (2005) note that, in the United States, nearly sixty percent of the country's imperiled species are found just outside the urbanized spaces of major metropolitan areas. Similar conditions can be observed in cities around the world, and, given current trends in land use change, the situation for these native species is not improving. Globally, urbanized land is expected nearly to triple during the next few decades (Seto et al. 2012), increasing its impact on habitat loss and the threat to global biodiversity.

Disruption of the Nitrogen Cycle

Nitrogen is a key nutrient in soil fertility; urbanization's impact on soil fertility, as mentioned above, was the primary motivation behind Karl Marx's notion of a metabolic rift (Foster 1999). In terms of the nitrogen cycle, just like biodiversity loss, urbanization’s impacts are both direct and indirect. These impacts involve not only the effects of agriculture on soils in rural areas producing the food and fiber for urban residents, but also the accumulation of nitrogen compounds in urban areas and in aquatic ecosystems connected to cities (Rockström et al. 2009; Vitousek et al. 1997). Therefore, the urban link to the nitrogen cycle is multidimensional, involving a complex of changes to global biogeochemical processes. Indeed, the current complexity of the relationship between urbanization and the nitrogen cycle has, in part, led some scientists to ask if modern cities have a distinct biogeochemistry because of the unprecedented accumulation of nutrients in urban areas (Kaye et al. 2006). Human waste and landscaping practices concentrate much reactive nitrogen in cities, which has a negative effect on aquatic ecosystems within and downstream from cities (Paul and Meyer 2001). Furthermore, with respect to the climate, the combustion of fossil fuel in cities results in nitrogen oxides (NO_x), thereby contributing to urban smog. Urban fossil fuel combustion also produces nitrous oxide, (N_2O), which is potent greenhouse gas.

Nevertheless, modern agriculture, mostly through its dependency on inorganic nitrogen fertilizer (Mancus 2007), is primarily responsible for the disruption of the nitrogen cycle. This process continues to introduce much new reactive nitrogen into the environment, "polluting waterways and the coastal zone, accumulating in land systems and adding a number of gases to the atmosphere" (Rockström et al. 2009). Agricultural soil management (e.g., the use of inorganic nitrogen fertilizer) is the single biggest source of anthropogenic N_2O, a greenhouse gas. Soil management also releases much reactive nitrogen into rivers and streams, contributing to eutrophication in bodies of water downstream from agricultural zones, ultimately threatening aquatic biodiversity as oxygen is removed from the environment (e.g., the dead zone in the Gulf of Mexico [Dybas 2005]). Again, the ecological consequences of modern agriculture may be observed in rural places, but they implicate urbanization and city-life indirectly. This rural-urban dynamic is the same basic tension that Marx was highlighting, again, in one of the first socio-ecological analyses of the modern society-soil metabolism.[3] He described this dynamic in terms of the town-country antagonism, which continues to characterize the relationship between rural agriculture and urban lifestyles today (Clement 2011; McKinney 2013). Indeed, in the 21st century, most humans have moved into cites and become socially and ecologically separated from agricultural production, the consequences of which, in terms of the nitrogen cycle, are multiple and far-reaching.

[3] Clearly, changes to the nitrogen cycle are far more complex than the science of Marx's time was able to understand. Nevertheless, some environmental social science scholars, while recognizing this history of intellectual and scientific development, fault Marx for conducting an "unnecessarily limited" socio-ecological analysis of soil fertility and management (Schneider and McMichael 2010: 469).

The Carbon Cycle and Climate Change

Climate change as a result of anthropogenic carbon emissions is perhaps the most discussed and most urgent environmental problem humans face today. Modern urbanization generates carbon dioxide, both directly and indirectly (Pataki et al. 2006). Directly, due to the fact that industry and transportation are concentrated in urban areas, "cities are point sources of CO_2 and other greenhouse gases" (Grimm et al. 2008: 757). In the United States, Parshall et al. (2010) find that American urban areas are responsible for up to 86% of all fossil fuel use by buildings and industry; they also consume up to 77% of total transportation fossil fuel. However, their data do not include "emissions associated with imported energy such as electricity" (4772), which they point out accounts for 40% of all direct energy consumption in the United States. Until recently, about half of electricity generated in the United States came from coal, and, as other studies have pointed out, urbanization increases the demand for energy (e.g., Clement and Schultz 2011; York, 2008); thus, "the exclusion of electricity understates total urban demand" of energy and production of carbon emissions (Parshall et al. 4772). Urbanization not only directly contributes CO_2 through the concentration of fossil fuel intensive activities, such as industry and transportation, but also indirectly produces carbon dioxide through the consumption of coal-based electricity, which is usually generated elsewhere. Combined, directly and indirectly, Grimm et al. (2008) estimate that 78% of global carbon emissions are attributed to cities.

Furthermore, through changes in land use, cities directly and indirectly contribute to the disruption of the carbon cycle (Seto and Shepherd 2009). For instance, urban population growth and the international demand for agricultural products (e.g., beef and soybeans) drive deforestation in tropical countries, a process that releases greenhouse gases as trees are cut down to make space for crops and pastures (DeFries et al. 2010). Deforestation not only releases the stores of carbon in the vegetation but also destroys important sinks for carbon uptake, which also threatens biodiversity as there is less net primary productivity (NPP), or carbon biomass (i.e., fuel for and food from plants and animals). The implications of land use change for the carbon cycle are also evident outside of the agricultural areas of tropical nations. While the built environment of cities represents a small fraction of total land area, urbanized spaces tend to occupy a disproportionately greater space of arable land with relatively high amounts of carbon biomass (Imhoff et al., 2004). In this way, urbanization also directly reduces NPP; the construction of impervious surfaces precludes the conversion of carbon biomass into fuel for plants and ultimately food for humans. For the United States, Imhoff et al. (2004) estimate the reduction of NPP due to urban land use to be about 1.6%. Because urbanization in the United States is happening on fertile agricultural land, this means that agricultural production is negatively affected. "In terms of actual human food," Imhoff et al., write, "the reduction of NPP from agricultural lands equates to food products capable of satisfying the caloric needs of 16.5 million people or about 6% of the US population" (442). In summary, urbanization drives land use changes through deforestation and the transformation of fertile lands into the built environment of cities. These forces work to destabilize the carbon cycle, which contributes to biodiversity loss and global warming.

The Environmental Challenge of an Unequal, Pro-Growth Society

Now that I have reviewed the local and global environmental consequences of urbanization and city life, I turn to a discussion of the prospects for developing sustainable solutions to these problems. From a social science perspective, some environmental scholars argue that urban sustainability is constrained by the forces of growth and inequality that characterize modern society (Agyeman et al. 2003; Pinderhughes 2004). Although particular propositions and terminologies may differ, prominent urban scholars generally agree that contemporary urbanization is marked by a high degree of inequality and is uniquely tied to the growth imperative and the various ways it manifests itself (Harvey 1982; Logan and Molotch 2007; Sassen 1991).[4] For instance, in the original development of growth machine theory, Molotch (1976) argued that growth is a "syndrome of associated events", which includes larger populations, an expanded labor force, greater economic activity, bigger governments, and "increasingly intensive land development" (310). According to this argument, land use intensification, as a dimension of urbanization, is fundamental to growth, and, as discussed above, it implicates critical changes to both the natural and built environments. Furthermore, the environmental changes wrought by growth are not experienced equally by all members of society; with unequal growth, some communities get protected parks and trees while others get dirty industries and pollution (e.g., Crowder and Downey 2010; Schwarz et al. 2015). This form of environmental inequality is what Molotch (1976) described as "aristocratic conservation", or the process by which people with greater political-economic power are able to use their communities "as a setting for life and work, rather

[4] See Conclusion for further discussion about how Logan and Molotch frame the broader political-economic context of "growth" versus "capitalism".

than as an exploitable resource" (328). Generally speaking, aristocratic conservation calls to mind similar concepts developed more recently by environmental social scientists, including "defensive environmentalism" (Rudel 2012a) and "environmental privilege" (Freudenburg 2005; Pellow and Brehm 2013). The basic argument underlining these different concepts concerns the ability of the dominant groups in society to preserve the unspoiled character of their local environments while displacing negative change onto communities with less political-economic power. It is in this context that I frame a discussion about the solutions to the major urban environmental problems facing humanity, focusing mostly on the environmental issue of population density but also briefly addressing some of the social science research on the connection between cities and green technologies.

In the socio-ecological literature on urbanization, population density is a commonly discussed topic; the various points made about density represent well the challenges posed by an unequal, pro-growth society. Furthermore, the issue of density has become a central point of debate in which opposing arguments are made concerning the environmental promises and pitfalls of urbanization and city-life (Satterthwaite 2008 and 2009). On the one hand, some scholars conceptualize urbanization in terms of population density and argue that higher density improves the efficiency of natural resource use, notably fossil fuel (Dodman 2009; Liddle 2013; Newman 2006; Owen 2009). On the other hand, a budding body of environmental scholarship emphasizes that urbanization is multidimensional (Elliott and Clement 2014) and that higher density, as one dimension of a larger process of change, may neither have a straightforward effect on natural resource use and pollution (Gately et al. 2015; Jones and Kammen 2014; Kaza, et al. 2011; Tamayao et al. 2014; Ummel 2014) nor generate the desired improvements in efficiency (Melia et al. 2011; Neuman 2006).

Meanwhile, social scientists have long agreed that higher population density is an essential feature of urbanization (Wirth 1938) and that increasing urban density makes a vital contribution to the various groups and institutions oriented around growth in modern society (Logan and Molotch 2007; Spence et al. 2009). Generally speaking, as the World Bank puts it, "Urbanization and growth go together" (Spence et al. 2009: 1). Or rather, the growth of modern societies has been characterized by an increasing proportion of the population living in urban, *not rural*, areas (Araghi 1995). And, to be clear, this transformation has been uneven, particularly as farming and agricultural communities around the world continue to be displaced by the unequal, pro-growth forces of the global economy (Araghi 2000). Meanwhile, as overall urbanization continues, many developed nations, and especially the United States, have experienced sprawling suburban development, which has resulted in a *relative* decentralization of an already mostly urban population. This relative decentralization, some scholars suggest, does not undermine the argument that growth is characterized by higher urban density (Anderson 1976; Logan and Molotch 2007). Anderson (1976), for example, writes, "Decentralization, outside of urban sprawl, is not profitable" (190).[5] Yet, many environmental social scientists focus

[5] Anderson (1976) speculates that that there is a limit to suburban sprawl, i.e., the degree of decentralization possible in a pro-growth society like the United States; in such a system, generally speaking, urban populations can only spread out to a certain extent. Recent trends in the U.S. provide corroborating evidence of this point. After World War II, the pace of urbanization in this country slowed down and, in the 1970s, even started to move in the opposite direction, with people returning to the countryside to live in rural areas (Johnson 2006). However, this urban-to-rural migration has not continued through the present, and, now, more than eighty percent of the nation's population lives in urban areas. Moreover, recent demographic analyses show that between 2010 and 2012 the total population of all rural counties, combined as a whole, declined for the first time since the Census started to collect and report county-level population figures (Cromartie 2013). Thus, recently, the relative *and* total size of the country's urban population has increased while the relative *and* total size of the rural population has declined. And, while

their critique on suburban sprawl, describing it as a decentralized, low-density urban form with a socio-physical structure that diminishes the efficiency of natural resource use (e.g., Gonzalez 2009). According to this argument, suburban sprawl drives negative environmental change as more land is eaten up by roads and buildings, and people have to drive farther and farther, both for work as well as for shopping and entertainment.

The various groups and institutions that support growth, including prominently, for example, the real estate and financial sectors, have profited enormously from the long history of car-centric suburban development in this country (Baran and Sweezy 1968; Hayden 2003). Logan and Molotch (2007) focus a substantial portion of their analysis on this history and also argue that suburban development has had a negative impact on the efficiency of natural resource use; they write, for instance, "Low-density and high-mileage commuting work forces drain resources in a way that tenement-living proletarians did not" (223). However, these authors simultaneously emphasize the connection between urban density and growth while considering issues of environmental change and social inequality. Broadly, they state, "Higher density has always been a scheme of growing rents; developers consistently lobbied for more on less. They didn't give a hoot about environment or social diversity" (xx). In this passage, Logan and Molotch are explicitly referring to alternative forms of residential development in the United States, including projects that advocate for higher densities such as "smart growth" and "compact development" (Duany et al. 2010; Ingram et al. 2009; Nielsen 2014). As they suggest, however, these alternative development strategies have not been consistently and successfully implemented across the United States (Knaap and Lewis

population density in the United States might continue to decrease due to suburban development, it will not reach the point where the country's residents could be considered rural.

2009). Meanwhile, the trajectory of social inequality, especially patterns of residential segregation in America's cities by class and race (Fry and Taylor 2012; Logan 2011), has remained largely unchanged. Additionally, smart growth and compact development plans have not been implemented on a scale capable of dealing adequately with the broader issue of the pollution generated and unequally distributed in cities. Indeed, the current level of residential segregation in the U.S. has structured the way the country's urban residents are variably exposed to pollution, with wealthier and whiter communities experiencing lower rates of exposure than other communities (Crowder and Downey 2010).[6] In this way, the success of these alternative development strategies will likely remain limited, having little effect on current levels of social and environmental inequality and posing little challenge to the pro-growth forces of modern society.

Although their implementation has been limited, many scholars argue that smart growth and compact development represent an advance in urban planning as long as they address issues of environmental inequality (Pinderhughes 2004). Meanwhile, there has also been continuous progress in the development of green technologies that could potentially ameliorate the environmental impacts of cities. Sassen and Dotan (2011), for instance, discuss a selection of technologies that have been designed specifically to address our urban impact on the nitrogen and carbon cycles; these include algal wastewater processing (which effectively takes nitrogen out of the wastewater and returns it to the atmosphere), algal fuel generation (sequesters carbon dioxide from the atmosphere to produce ethanol), self-healing concrete (improves the durability of the

[6] In recent research, urban scholars have made a distinction between exposure to toxins that are known and/or officially reported versus exposure to the hazardous wastes generated from past urban industrial activities that were covered up as land use changes over time (Elliott and Frickel 2013). In this research, income and race do not influence exposure to the toxic wastes that have been historically obscured.

built environment), and bio-reactor landfills (can turn the gases generated from waste decomposition into usable energy). If widely implemented, these and other renewable technologies would help to reduce the use of natural resources and help to recycle the pollution generated in urban areas. Yet, as Sassen and Dotan acknowledge, the implementation of these green technologies, like the alternative development strategies discussed above, remains limited. Furthermore, empirical analyses looking at the connections between renewable energy technologies, urbanization, and growth have questioned their efficacy in terms of reducing overall environmental impact. For instance, in cross-national panel studies, Salim and Shafiel (2014) do not find any effect of urbanization on the use of renewable energy, and York (2012b) shows that, as renewable energy technologies were increasingly deployed over time, there was not a proportionate reduction in the use of the more carbon intensive energy sources they were intended to replace.

Based on his findings, York argues that a singular focus on technological development will at best only partially address the environmental impacts of modern society; a more complete and comprehensive approach, he argues, would also consider the various pro-growth forces that encourage and rely on the increasing use of natural resources. Indeed, the issue of growth is now being questioned in the natural science community, perhaps most prominently in the *People and the Planet* report published by the Royal Society of London (2012). In this report, a team natural scientists highlights, among other problems, the urban environmental challenge posed by an unequal, pro-growth society, also noting the proliferation of slum life and the high levels of poverty that persist around the world. These scientists argue that, alongside technological developments and better urban planning, the national governments of the world must collaborate to develop "socio-economic systems and institutions that are not dependent on continued material consumption growth" (109). What these scientists are saying is that human society must abandon growth as a primary goal that motivates not only the decisions and activities of the various institutions in modern society but also our entire socio-economic system. Challenging growth and inequality, in combination with better planning and greener technologies, constitute a more comprehensive approach to urban environmental problems. In fact, without this more comprehensive approach, the report claims, the prospects of urban sustainability are unclear.

As our urbanized society approaches a critical juncture in its changing relationship with the biosphere (Grimm et al. 2008), there is a pressing need to move in a more sustainable direction. Despite marginal improvements in the use of natural resources, increasing the density of our cities and deploying greener technologies do not represent sufficient steps in the direction of sustainability, especially in terms of biodiversity loss as well as the disruption of the nitrogen and carbon cycles. For instance, higher density, in and of itself, does not change the way food and fiber are being produced in and extracted from rural areas. Thus, without a transformation in agricultural and forestry practices, environmental degradation in rural areas will carry on due to the intensive use of inorganic fertilizers as well as the destruction of habitats and carbon sinks, ultimately for the purposes of feeding, clothing, and housing urban residents. In fact, given current practices, increasing urban density might even exacerbate these rural impacts, as more humans will become socially and ecologically separated from agricultural production and forestry. In that way, the prospects for urban sustainability are at best uncertain, and this prognosis will not change if other socio-ecological issues are missing from the discussion of urban sustainability, including a broader transformation of the unequal, pro-growth forces of modern society. Such a transformation and its implications for urbanization and city-life will pose a separate host of questions for environmental social scientists. To be clear, I do not pursue that line of inquiry in the subsequent chapters of my dissertation; instead, I identify and analyze the socio-ecological dimensions and consequences of urbanization at the local level across the United States. In this light, I present my dissertation as necessary groundwork for future socio-ecological research on potential alternatives to the unequal, pro-growth forces that characterize modern society in the U.S. and around the world (cf. Odum and Odum 2001). Meanwhile, the results of the quantitative analyses in the subsequent chapters demonstrate that urbanization entails more than just changes in population density. In the conclusion, I summarize the findings of these analyses and discuss their policy implications given the overwhelming focus on density in the current discussion about urban sustainability.

CHAPTER II

THE ENIRONMENTAL CONSEQUENCES OF RURAL AND URBAN POPULATION CHANGE: AN EXPLORTATORY SPATIAL PANEL STUDY OF FOREST COVER IN THE SOUTHERN UNITED STATES

A revised version of this chapter was published in *Rural Sociology* with Christina Ergas and Patrick Trent Greiner as co-authors. My co-authors helped frame the argument, write the section on ecological modernization, edit the paper, and locate references on forest cover change in the Southern United States.

Forest cover change is a fundamental human activity (Chew 2001; FAO 2012; Williams 2010) and, as mentioned in the previous chapter, is implicated in a number of socio-environmental issues, including global warming, appropriation of net primary productivity, habitat loss, the spread of invasive species, wildfires, landslides, flooding, forestry, agriculture, biofuels, recreation, sprawl, among other concerns (Cramer and Hobbs 2007; Egan and Luloff 2000; Ellis 2011; Holleman 2012; Krausman et al. 2013; MacDonald and Rudel 2005; Miller 2012; Neumann 2007; Perz 2001; Riall 2007; Walker and del Moral 2003). While several sociologists have contributed to this diverse literature on forest cover change, researchers in the discipline have had a particular focus on the systemic causes of deforestation, using quantitative analysis to examine the demographic and economic drivers behind this type of anthropogenic impact (e.g., Austin 2010a; Burns et al. 2003; Ehrhardt-Martinez 1998; Jorgenson 2006; Jorgenson and Burns 2007; Rudel 1989; Shandra et al. 2012). Furthermore, within this literature, there has been a fair amount of attention paid to the relative effects of rural and urban population change, either as variables of primary theoretical interest or as important controls. While the findings from this research have changed as the quality of forest cover data has improved, an issue I will discuss below, the vast majority of these studies have been conducted at the national level, with negligible attention paid to local-level dynamics (for an exception see MacDonald and Rudel 2005).

The second chapter in my dissertation aims to fill this gap in the literature, conducting a local-level analysis of the relative effects of rural and urban population change on total forest area across the Southern United States between 2001-2006. The Southern U.S. at the start of the 21st century represents an ideal context in which to examine the systemic drivers of change in total forest area. I will discuss this context in greater detail below; for now I highlight that not only does the Southern U.S. currently contain a significant portion of the world's forest area, but recently it also has experienced a high degree of rural and urban population change as well as deforestation *and* afforestation (Hanson et al. 2010). On that note, the following analysis makes two general contributions to the literature: 1) it contributes a local-level perspective to a theoretical debate largely taking place at the national level concerning the relative environmental effects of rural and urban population change, and 2) it uses a spatial panel model to test these effects. A spatial panel model incorporates spatial effects into a regression analysis using longitudinal data, thereby addressing the issue of spatial dependence in repeated observations over time for areal units (Lesage and Pace 2009). This type of panel model represents an advance in the methodology commonly used in quantitative studies of deforestation, which frequently controls for temporal dependence (e.g., Austin 2010a) and only rarely controls for spatial dependence (e.g., MacDonald and Rudel 2005), but not both.[7]

To that end, the chapter proceeds through the following steps. First, I report changes in forest cover in the Southern U.S. at the start of the 21st century, highlighting the utility of this context to analyze rural and urban population change as systemic drivers of deforestation and afforestation. Second, I review literature in sociology to discuss the effects of rural and urban population change on the natural environment, with a particular emphasis on forest cover. In this review, I frame the competing arguments about these effects in terms of the debate between ecological modernization theory (EMT) and urban political economy (UPE). Third, I describe the data and the analytic technique used to test hypotheses based on the literature review. The analysis combines demographic and economic indicators from U.S. government sources with data on total forest area from the National Land Change Database (NLCD) (Fry et al. 2011). Fourth, I report and discuss the results from spatial panel models, providing a comparison with results from conventional panel models that do not control for spatial autocorrelation. Lastly, in the conclusion, I elaborate on the ways that rural and urban population change might be differentially related to forest cover in the Southern U.S. context.

[7] Unlike other social scientists (e.g., Liu et al. 2014), sociologists in general have not readily incorporated spatial procedures into panel models. In fact, even when there are repeated observations over time for areal units, the longitudinal structure of the data is avoided with the use of change-scores, which are then incorporated into a conventional spatial regression model (e.g., Elliott and Clement 2014; Genter et al. 2013).

Forest Cover and Rural/Urban Change in the Southern United States

The Southern United States occupies an area of roughly 885,000 square miles, on which sits a little more than 280,000 square miles of forests, representing nearly 32% of the region's area and, according to Hansen et al. (2010), about two percent of total global forest area. Based on data from the NLCD (Fry et al. 2011), all four U.S. Census regions (Northeast, Midwest, South, and West) experienced net deforestation between 2001-2006; nonetheless, the South had the largest amount of forest loss *and* gain (i.e., deforestation and afforestation), both in terms of total square miles and as a fraction of total area (see Figure 1).[8] In the Southern U.S., the total area covered by forests declined from 284,698 square miles in 2001 to 281,199 square miles in 2006, a net loss of almost 3,500 square miles. However, this net loss obscures the total amount of land transformed during this time, in which 11,310 square miles of trees were cut down and 7,810 square miles of land was forested. Thus, a little more than 19,000 square miles of land experienced either deforestation or afforestation, an area roughly the size of Costa Rica.

Yet, within the South, rates of de/afforestation were not evenly distributed across space. Figure 3 displays percent change in forest cover at the county-level across the Southern U.S. for the period 2001-2006. During this time, 1,064 Southern counties (nearly 75% of the sample) experienced net deforestation, 336 counties experienced a net increase in forest cover, and the remaining 23 counties experienced no change at all. Based on the Moran's I, a measure of spatial autocorrelation, there is significant clustering of forest gain and loss at the county-level during this time (I=0.296;p<0.001).

[8] I use the term "afforestation" to refer to forest gain. The NLCD data are based on two waves of satellite imagery, one from 2001 and one from 2006 (more details below); as a consequence, I use the term "afforestation" rather than "reforestation" because one cannot tell whether or not any of the forest gain had taken place on land that was forested prior to 2001. Alaska and Hawaii are not included in the West region.

For instance, counties that experienced afforestation tended to cluster in the west and southern tip of Texas; parts of Louisiana and Arkansas, and western Mississippi; southeastern Alabama, southern Georgia, and South Carolina; as well as the Chesapeake Bay area. Counties that experienced the greatest forest loss tended to cluster in southeastern Texas, along the border of Mississippi and Alabama, and along the coast of North Carolina. Outside of the notable areas of clustering, there were other hotspots with very high rates of forest loss, including, for example, the corridor between Bexar and Travis Counties in Texas (where the cities of San Antonio and Austin are located, respectively) and the greater metropolitan areas of Atlanta in Georgia and Orlando in Florida.

Of the four U.S. Census regions, the South not only has the largest area of de/afforestation, it also witnessed the greatest change in rural and urban populations, in both absolute and relative terms (see Figure 2). In the South, between 2001-2006, while the number of people living in rural areas increased by roughly 224,000, the percent rural declined by nearly 1.5%. Thus, while the South's rural population grew, its urban population grew faster. None of the other three Census regions experienced an equivalent level change, in either absolute or relative terms. Nevertheless, an aggregate summary of rural and urban population change for the entire Southern U.S. masks the wide variation evident at the county-level. For instance, while the rural population of Wake County, NC declined by nearly 10,000 residents, the number of people living in rural areas in St. Johns County, FL increased by approximately 8,300; the size of St. Johns' rural population in 2006 was 1.73 times its size in 2001. Similarly, while 794 counties in the

Figure 1. Relative and Absolute Area of Forest Gain and Loss by U.S. Census Region, 2001-2006 (Forest gain in grey; forest loss in black).

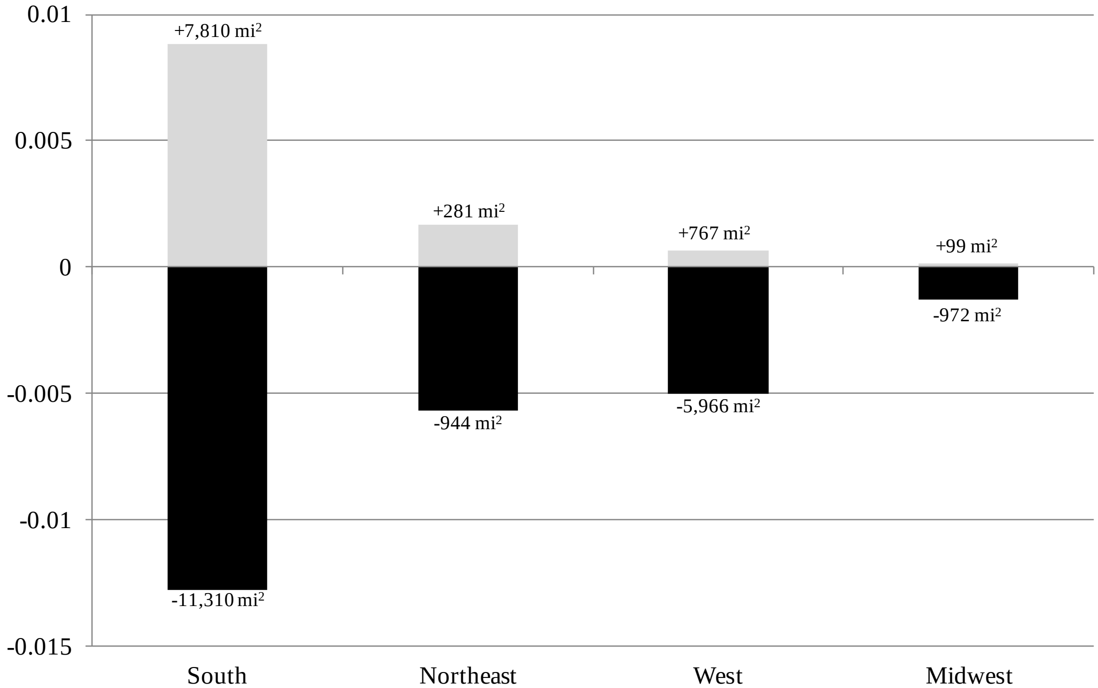
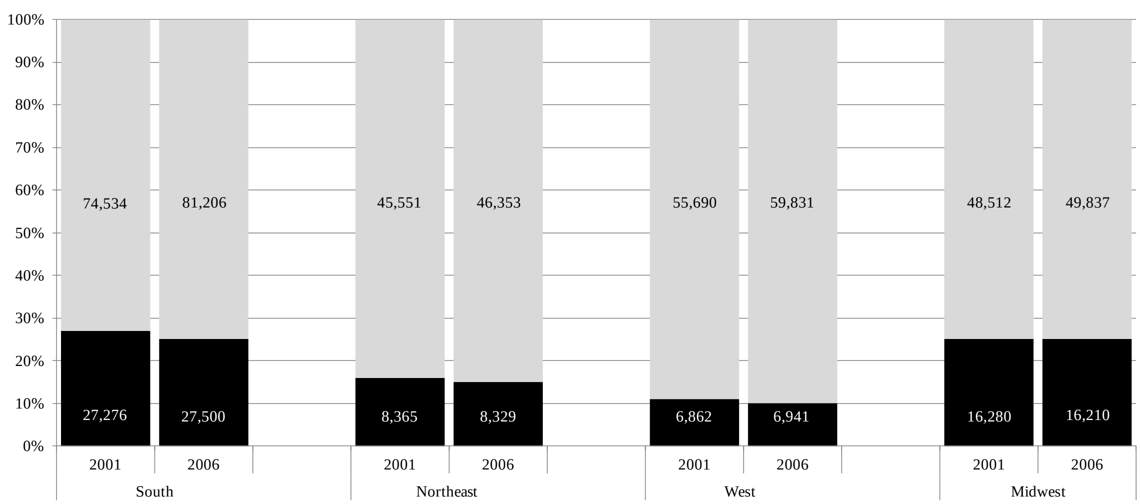


Figure 2. Relative and Absolute Size of Rural and Urban Population by U.S. Census Region, 2001-2006 (Urban population in grey; rural population in black).



Note: The bars represent the percent of the total population living in rural and urban areas. The numbers inside the bars indicate population size (in thousands).

South experienced urban population growth, 365 counties saw their urban populations decline and 264 counties had no change at all (see Figure 4).

In summary, the absolute scale and relative degree of change highlight the utility of the Southern U.S. as an ideal context for examining connections between rural/urban population size and forest cover at the county-level. But, how have sociologists theorized rural/urban population as a systemic driver of environmental change, in general, and de/afforestation, in particular? Most of this work has happened at the national level; thus, while I have demonstrated the importance of the Southern U.S. as a context for examining this socio-ecological process at the local level, I recognize that most theorizing concerns a higher level of analysis. Consequently, I highlight the exploratory character of this chapter but argue that the hypothesized mechanisms through which rural and urban population size drive forest cover change can be tested at the sub-national level.

Literature Review and Theory

There has been much sociological research examining environmental change in rural areas (for overviews see Albrecht and Murdock 2002; Field and Burch 1988). A good portion of this has been concerned with agricultural practices, forestry, mining, and recreation; indeed, discussion about rural environmental problems tends to focus on the extractive nature of rural economies. Buttel (1996) credited rural sociology's focus on environmental and natural resource issues as having an influence on the development of environmental sociology. Thus, even though environmental sociologists had long

Figure 3. Map of Percent Change in Forest Area, Southern United States, 2001-2006 (Dark shading indicates positive values; no shading indicates negative values).

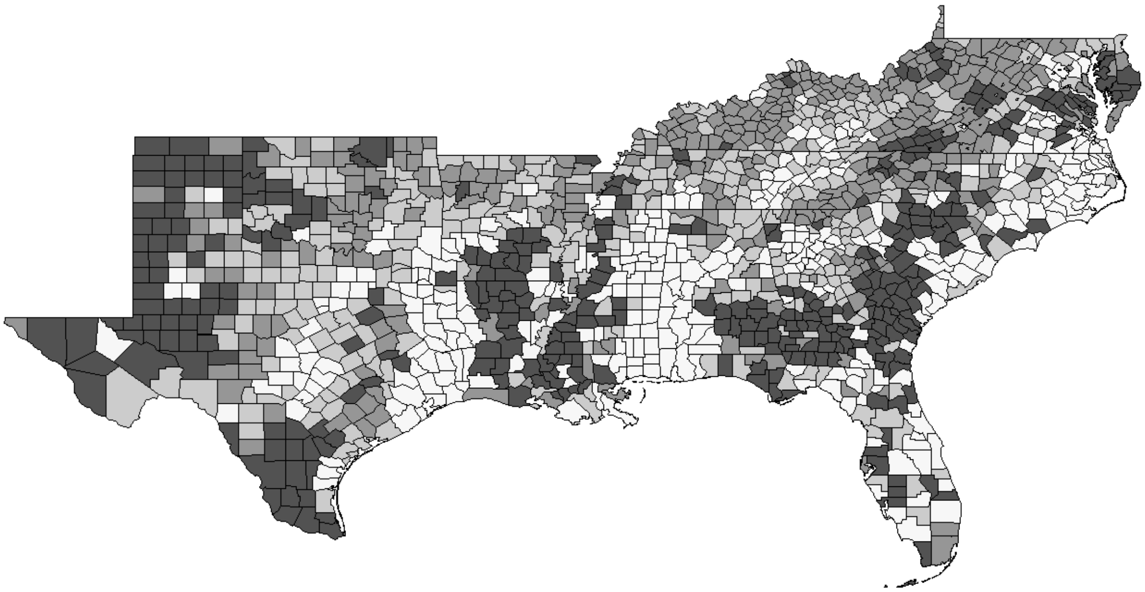
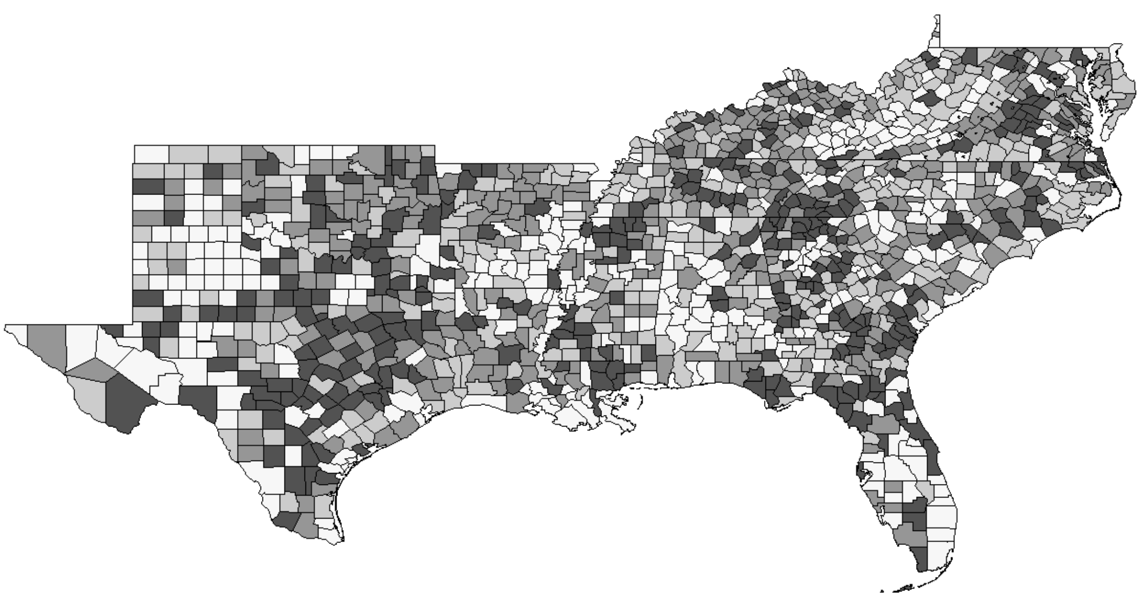


Figure 4. Map of Percent Change in Rural Population, Southern United States, 2001-2006 (Dark shading indicates positive values; no shading indicates negative values).



recognized the environmental consequences of urban areas (e.g., Anderson 1976; Catton 1980), more attention had been focused on the connection between rural areas and environmental change. More recently, however, this attention has shifted; there have been several theoretical and empirical studies examining the links between urbanization, city life, and the natural environment (e.g., Chew 2001; Clement 2010; Elliott and Clement 2014; Ergas 2010; Jorgenson et al. 2010; Taylor 2009; Weinberg et al. 2000; Wachsmuth 2012). While these authors ask a variety of research questions, one question in particular has motivated a number of studies, this question is: What are the relative ecological impacts of rural and urban population growth? Empirically, this question has been the focus of discussion in quantitative studies of deforestation (DeFries et al. 2010; Ehrhardt-Martinez 1998; Ehrhardt-Martinez et al. 2002; Jorgenson and Burns 2007; MacDonald and Rudel 2005; Rudel 2013). As a whole, this literature draws from different theoretical perspectives, presenting competing arguments that suggest rural and urban population growth are differentially related to changes in forest cover. At their root, these competing arguments involve the debate between ecological modernization theory (e.g., Ehrhardt-Martinez 1998) and urban political economy (e.g., Burns et al. 2003). I now review the EMT and UPE arguments as a way to develop hypotheses that I then test at the county-level in the Southern U.S., again an area that has experienced much change in both rural/urban population size and area of forest cover.

Ecological Modernization vs. Urban Political Economy

The literature on deforestation has reported three different findings with respect to rural/urban population size and area of forest cover:

1) urban growth is associated with afforestation (e.g., Ehrhardt-Martinez 1998; Ehrhardt-Martinez et al. 2002; Jorgenson 2006; Jorgenson and Burns 2007; Rudel 1998),

2) rural growth is associated with deforestation (e.g., Jorgenson 2006; Jorgenson and Burns 2007; Shandra et al. 2009; Shandra et al. 2011; Shandra et al. 2012), and

3) urban growth is associated with deforestation (e.g., Burns et al. 2003; DeFries et al. 2010; Rudel 2013).[9]

The first two findings are compatible and are seen as part of the same theoretical framework: rural population growth is detrimental to the environment, whereas urban population growth is beneficial. These processes are generally framed as support for EMT, with urbanization being a principal indicator of modern development that relieves anthropogenic pressures on the natural environment (Ehrhardt-Martinez 1998; Ehrhardt-Martinez et al. 2002). In contrast, the third finding implicates the UPE framework, which emphasizes the negative environmental consequences of urbanization (Jorgenson and Clark 2011).

On the one hand, EMT argues that the "path to sustainability lies in understanding the [modernization] process" (Scheinberg and Anschütz 2006: 268). Mol (2002) describes how environmental thinking and stewardship emerged in modern societies in the 1970s and have since permeated governmental and economic policies. As a result,

[9] There are a few qualifications to note in this summary. For instance, Burns et al. (2003) find that the detrimental effect of urbanization is most pronounced in semi-peripheral and peripheral nations. Ehrhardt-Martinez (1998) and Ehrhardt-Martinez et al. (2002) find an environmental Kuznets curve between urbanization and deforestation. And, several studies have found no significant effect of either rural or urban population (e.g., Austin 2010a; Austin 2010b; Austin 2012; Shandra 2007).

Mol argues, these activities have become institutionalized, ensuring the "permanence" of ecologically-sensitive practices. In this light, the environmental consequences of human actions are generally taken into account through the ongoing development of various modern institutions and processes (see Mol 2002: 94). For instance, in much of the early work on EMT, discussion about the modernization process had focused on the dematerialization of economic growth, that is: "Environmental improvement can go together with economic development via a process of delinking economic growth from natural resource inputs and outputs of emissions and waste" (Mol 1997: 141). Although modernization scholars outside of the environmental literature had previously emphasized the connection between urbanization and development (Kasarda and Crenshaw 1991), the environmental implications of this connection were not adequately scrutinized until Ehrhardt-Martinez (1998) and Rudel (1998).[10] Nevertheless, these and other studies treat urbanization as a proxy for the degree of industrialization in an economy or the type of fuel used in the economy. That is, as the rural population declines and the urban population grows, the number of farms and the level of agricultural activity decline, which reduces the pressure placed on forest resources, helping to slow down deforestation. Similarly, according to this argument, urbanization is also related to technological innovation. Among other things, this innovation means wood products are replaced with fossil fuels, which also helps to relieve pressure on forest resources. In summary, according to EMT, urbanization is beneficial for the environment.

[10] Citing unpublished results, Crenshaw and Jenkins (1996) describe a set of propositions about the effect of urbanization on greenhouse gas emissions, hypothesizing that urban agglomeration improves the efficiency of fossil fuel use, thereby helping to reduce greenhouse gas emissions. In contrast, Ehrhardt-Martinez et al. (2002) argue that advanced urbanization is "characterized by…increased use of petroleum, coal, and electricity" (229), thereby intensifying the human production of greenhouse gases.

On the other hand, according to the UPE framework, rural and urban areas are involved in an unequal economic exchange (Lobao et al. 2007; London and Smith 1988), which takes an ecological form, with the latter treated as a supply depot and repository for urban activities (Burns et al. 2003; Buttel and Flinn 1977; Lichter and Brown 2011). To make this point, environmental sociologists draw on growth machine theory (Molotch 1976), arguing, for example: "Cities remain centers of growth that require massive amounts of natural resources to sustain daily operations" (Jorgenson and Clark 2011: 240). According to growth machine theory, urban-based political and economic elites strive to extract more and more exchange value through land development projects. These elites also encourage the expansion of commodified activities that generate revenue for themselves and their shareholders. In environmental terms, this heightened level of human activity amplifies the socio-ecological metabolism of urban areas, increasing the use of natural resources. In terms of forest resources, this process generates an unequal rural-urban exchange, which calls to mind metabolic rift theory (Burns et al. 2003; Foster 1999). Indeed, DeFries et al. (2010) argue that urbanization raises consumption levels, which means that more trees are cut down in rural areas to make space for commercial food production, which then feeds residents in urban areas. Rudel (2013) makes a similar argument; agricultural production, which contributes to deforestation, is driven by urbanization, which creates a demand for a diet richer in animal products (cf. York and Gossard 2004). Thus, urbanization is treated as a proxy for the expansion of farmland. Meanwhile, this research acknowledges that low-income nations are generally net-importers of food (Ng and Aksoy 2008; Rakotoarisoa et al.

2011; Rudel 2013) but argues that domestic agricultural production is directed towards the *domestic* urban market.

Exploratory Hypotheses for a Local Level Analysis in the Southern U.S.

The majority of the research examining the drivers of forest cover change has been conducted at the national level. Nevertheless, in terms of the effects of rural and urban population change, the mechanisms described in this research are not exclusive to that level of analysis. I argue that the same propositions about rural/urban change can be applied to the sub-national level. Indeed, these researchers talk about urbanization in terms of the level of industrialization, the presence of primary production, and the socio-ecological metabolism, all three of which are characteristics of urbanization that can be operationalized at the regional, state, and county levels. Furthermore, the effects of rural/urban population change have been examined in a variety of contexts, including both developed and less developed nations. Thus, the present exploratory chapter contributes a local-level evaluation of a theoretical debate that has taken place largely at the national level. Based on EMT and UPE, I test the same hypothesized mechanisms through which rural and urban population are said to drive changes in forest cover. These hypotheses are as follows:

H_1: *urban population size is positively associated with forest cover*

H_2: *rural population size is negatively associated with forest cover*

H_3: *urban population size is negatively associated with forest cover*

Nonetheless, these three propositions are not exhaustive of the different possible ways that rural/urban population growth could potentially drive forest cover change. In particular, this list leaves out the possibility that *rural population size is positively associated with forest cover*. There are two points to make with respect to this hypothesis. First, it is not compatible with EMT. According to EMT, rural population growth would be a reversal of the modernization process, so rural population size should be negatively related to forest cover: as the size of the rural population declines, the area covered by forests should increase. Likewise, as the rural population increases, which is opposite to the trend that characterizes the modernization process, the area covered by forests should decline. Second, a positive association between rural population size and forest cover would be the converse to the third hypothesis, which is based on urban political economy. The UPE argument, however, has focused specifically on the way urbanization amplifies the socio-ecological metabolism, which indirectly drives changes in forest cover through the expansion of remote farmland. This framework has not examined the converse process, that rural life moderates or restrains the socio-ecological metabolism, thereby limiting environmental impact. Thus, for a more exhaustive list of hypotheses, I include the following fourth proposition in this exploratory analysis:

H_4: *rural population size is positively associated with forest cover*

Data and Methods

Dependent Variable

The data for the dependent variable in this chapter come from the National Land Cover Database (NLCD), which is published by the Multi-Resolution Land Characterization consortium (MRLC) (Fry et al. 2011). The MRLC is composed of members from the following ten different federal agencies: the US Geological Survey, NASA, the EPA, NOAA, the US Forest Service, Bureau of Land Management, National Park Service, US Fish and Wildlife Service, US Army Corps of Engineers, and the National Agricultural Statistical Service of the USDA. Using satellite imagery from the Landsat program, this research collaboration has produced GIS raster data on land cover for the entire continental United States at a resolution of 30 X 30 square meters for the years 2001-2006. Of the sixteen different types of land cover identified in the NLCD, there are three categories for forest cover: "deciduous", "evergreen", and "mixed forests". According to the NLCD, forests are "areas dominated by trees generally greater than 5 meters tall and greater than 20% of total vegetation cover." The distinction between the three types of forest cover depends on whether or not most of the trees in the 30 X 30 square meter parcel shed leaves seasonally. If at least 75% of the parcel is covered by trees that shed foliage seasonally, it is designated deciduous. If at least 75% is covered by trees that hold onto their leaves annually, the parcel is designated evergreen. And, if neither deciduous nor evergreen make up at least 75%, then the parcel is considered mixed forest.

Using the Zonal Tabulate tool in ArcGIS, I quantified the area covered by deciduous, evergreen, and mixed forest for all 1,423 counties in the Southern U.S. Next, the areas covered by each type were summed up, yielding total forest cover at the county-level. This procedure was done for both 2001 and 2006, giving a total sample size of N=2,846 county-years. Next, to normalize the data, I log-transform the values of the dependent variable; I discuss other reasons for the log-transformation below in the Methods section.[11]

[11] The type and quality of data used in this analysis are comparable to what have been used in previous cross-national analyses of the drivers of deforestation using satellite imagery (DeFries et al. 2010; Rudel

Independent Variables

There are two primary independent variables and several controls incorporated into the regression analysis. The data for these predictors come from several U.S. government sources. Controlling for *total population size*, the two primary independent variables are *rural population size* and *urban population size*. These variables are based on data from the U.S. Census. *Total population* is the total number of people residing in a county; *urban population* is the number of people residing in Census-defined urban areas, which include all incorporated places with at least 2,500 residents; and *rural population* includes all people not living in Census-defined urban areas (for discussion about urban/rural designations see U.S. Census 2013). Rural and urban populations are counted in the decennial census. Thus, values for 2001 and 2006 were linearly interpolated using the 2000 and 2010 censuses. To normalize the data, all three primary predictors were logged transformed. Lastly, I also create interaction terms between rural population size and a dummy-variable for state which I use to examine whether the effect of the main variable varies across space.

For theoretical and methodological reasons, the following eight control variables are incorporated into the analysis: *number of farms*, *number of forestry operations*,

2013). Here, I address three issues regarding the limitations of these data. First, while cross-national data report forest cover change between 2000-2005, the present chapter examines the period between 2001-2006. Second, cross-national deforestation data based on satellite imagery distinguish neither between deciduous and evergreen forests nor between managed and unmanaged forests. Consequently, as DeFries et al. (2010) acknowledge, these data may include changes due to tree plantation harvesting and regrowth and do not differentiate between primary and secondary forests as well as different species (181). Third, based on the NLCD definitions, only four counties in the Southern U.S. had no forest cover in 2001 and 2006. Three of these counties (Cochran, Loving, and Winkler) are located in the dry climate of West Texas and the Texas Panhandle. The fourth county (Monroe) contains both the Florida Everglades and the Florida Keys. According to NLCD criteria, Monroe is covered by woody and herbaceous wetlands, not by forests. While I recognize these data limitations, I maintain their use for comparability with previous research; nevertheless, I also incorporate controls and other methodological strategies to address these limitations directly.

median household income, total payroll, *percent white*, *percent elderly*, *percent forest area in 2001*, and a *dummy-variable for year*. Data for these variables come from the following sources: US Census, Census of Agriculture, and the USDA's Economic Research Service. *Number of farms* is equal to the number of business operations that produce and sell at least $1,000 of agricultural products.[12] *Number of forestry operations* equals the total number of business establishments in a county that grow and harvest timber. These first two variables control for the organization of the local economy, as both agriculture and forestry are land-based activities that have been implicated in forest cover change (Befort et al. 1988; Egan and Luloff 2000; Neuman et al. 2007; Perz 2001). *Median household income* is equal to the county's median household income. *Total payroll* is equal to the total private non-farm payroll in a county, which includes all forms of compensation, such as salaries, wages, and officer and executive pay, among other items. Based on growth machine theory (Molotch 1976) and treadmill of production theory (Schnaiberg 1980), these two variables control for the effects of residential affluence and the size of the local economy on forest cover. *Percent white* is the percent of the total population that identifies their racial category as "white". Previous research has indicated that whites live in areas with more tree cover and non-whites in areas with less tree cover (Harlan et al. 2008; Jesdale et al. 2013). *Percent elderly* equals the percent of the population aged 65 years or older and controls for the potential effect of an aging population (Luloff and Krannich 2002), which includes the development of retirement communities and amenity migration (Egan and Luloff 2000; Gosnell and Abrams 2011). *All else equal*, these processes should contribute to deforestation. *Percent forest cover in*

[12] Using data from the 1997, 2002, and 2007 Censuses of Agriculture, values for *number of farms* in 2001 and 2006 were linearly interpolated.

2001 is the percent of the county's total area covered by forests in 2001; this predictor controls for the possibility of floor/ceiling effects (Firebaugh and Beck 1994); counties with a large area of forest cover have more to lose compared to counties with little forest cover, and vice versa. Lastly, the inclusion of a *dummy-variable for year* incorporates a fixed-effect for time (Allison 2009). With the exception of the dummy-variable for year, all predictors have been logged-transformed.

Methods

Area of forest cover is regressed on the independent variables in four different models. Because the dependent variable and the predictors have all been logged, the slope estimates from the regression models are interpreted as elasticities, representing the percent change in forest cover for every one percent change in the predictor, holding the rest of the equation constant. This procedure not only yields standardized (and thus comparable) estimates, but it also situates this chapter within the broader STIRPAT research program (Dietz and Jorgenson 2013; York and Rosa 2012). STIRPAT is a regression model that can be used to evaluate competing theoretical frameworks on the systemic drivers of environmental change.

The slopes are estimated using a conventional panel model and a spatial panel model (Belotti et al. 2013; see also Lesage and Pace 2009), both with two-way fixed effects. In a spatial panel model, as with a conventional panel model, not only can I examine change over time while controlling for temporal effects (Allison 2009), but I can also incorporate additional controls for related issues with respect to space, which is a concern when using an areal unit of analysis (Anselin and Bera 1998). With this procedure, the regression model controls for spatial autocorrelation in the dependent variable for each time period. Without controls for spatial autocorrelation, regression analysis not only violates the assumption of independent observations but also runs the risk of producing deflated estimates for the standard errors, which could yield overly generous results for significance tests (Anselin 2002). For this chapter, I incorporate panel data into a spatial autoregressive model with spatial autoregressive disturbances, known as a SARAR or SAC (spatial autocorrelation) model. This form of spatial regression incorporates both a spatial lag and a spatial error term, which control for spatial clustering not only in the values of the dependent variable but also in the residuals (Anselin and Florax 1995).

With two-way fixed effects, the generic equation for this type of spatial panel model is as follows:

$$y_{it} = \alpha + \rho W y_{it} + x_{itk}\beta_k + \upsilon_{it}$$

$$\upsilon_{it} = \lambda W \upsilon_{it} + \varepsilon_{it}$$

The symbol α is the constant, y_{it} indicates the values of the dependent variable for the i^{th} case at time t, and x_{itk} indicates the value of the k^{th} predictor for the i^{th} case at time t, with β_k representing the effect of the k^{th} predictor on the dependent variable. The spatial lag term ρ represents the weighted effect of the values of the dependent variable in neighboring units on the values of the dependent variable for the i^{th} case. This weighted effect ρ is based on the spatial weights matrix W. In this chapter, since county borders did not change between 2001-2006, W is the same for all t; it is a row-standardized, first-order queen contiguity spatial weights matrix, where the weight equals "1" for any county that touches the i^{th} case and "0" otherwise. Thus, the spatial lag for the i^{th} county at time t

is equal to the average forest area at time *t* for all of the counties that immediately border the i^{th} case. The error term υ_{it} is decomposed into two parts. The first part estimates the spatial error term λ, which is based on the same contiguity weights matrix *W*, and the second part ε_{it} represents all the left-over unobserved variation in the dependent variable. This estimation procedure was carried out in *Stata* using the command "xsmle" (Belotti et al. 2013).

Results and Discussion

Table 1 reports the summary statistics for the primary predictors and seven controls for the two years of data (2001 and 2006) in addition to their change scores during this time interval. Because all variables have been logged, the change-scores are measures of proportional change. For instance, the mean value for the change-score for *forest area* ($\bar{x}$=-0.009;p<0.001) indicates that the average area covered by forest declined by nearly 1% between 2001-2006. With the exception of *forest area*, *forestry operations*, and *percent white*, the change scores for all variables are positive, indicating that their

Table 1. Summary Statistics (All variables have been log-transformed) (N=1,423).

	2001		2006		Δ between	
	Mean	SD	Mean	SD	2001-2006	
Forest Area	4.621	1.641	4.612	1.634	-0.009	***
Urban Population	7.785	4.037	7.871	4.045	0.085	***
Rural Population	9.405	1.483	9.413	1.481	0.008	**
Total Population	10.262	1.228	10.289	1.263	0.027	***
Number of Farms	6.158	0.957	6.209	0.912	0.051	***
Forest Operations	1.167	0.028	1.059	0.027	-0.108	***
Median Household Income	10.507	0.239	10.510	0.244	0.003	**
Total Payroll	18.945	1.955	19.018	1.976	0.073	***
Percent White	4.353	0.279	4.348	0.278	-0.005	***
Percent Elderly	2.615	0.274	2.643	0.258	0.027	***

*p<0.05; **p<0.01; ***p<0.001 (one-tailed paired t-tests) mean values increased during this time interval. Table 2 reports the results from the panel and spatial panel models with two-way fixed effects. I discuss these results in four steps. First, I compare estimates from the panel and spatial panel models. Second, I discuss the effects of urban and rural population size. Third, I examine variation in the effects of rural population size across space. Fourth, I briefly report results for control variables.

First, as previously noted, the area of forest cover is spatially autocorrelated at the county-level. Thus, the estimates from a conventional panel model are unreliable because it assumes independent observations. Looking at the models in Table 2, I report evidence that spatial autocorrelation is affecting the results; indeed, the results change when I control for spatial autocorrelation. Models 1 and 3 do *not* control for spatial autocorrelation; Models 2 and 4 do. Comparing Models 1 and 2, the estimate for *population size* is only marginally significant and the estimate for *total payroll* becomes significant after controlling for spatial autocorrelation. Comparing Models 3 and 4, one notices that the main effect of *rural population size* and the estimate for *total payroll* become significant whereas *percent elderly* is no longer significant. In the spatial panel model, one also sees that the interaction terms between *rural population size* and state become marginally significant and significant, respectively, for *South Carolina* and *Virginia*. While there are also differences in the magnitudes of the slope estimates, here I focus on their significance levels to emphasize the need to incorporate spatial controls into longitudinal regression analyses.

Second, looking at Model 2, I discuss the estimates for urban and rural population size. While the former is positive but not significant, the latter is positive and significant (b=0.024;p<0.01).[13] For every one percent change in rural population size, the area of forest cover changes in the same direction by 0.024 percent. Controlling for *total population*, this result indicates that *rural population size* has an independent effect on forest cover but *urban population size* does not and that the effect of *rural population size* is not in the direction hypothesized by EMT. That is, the size of the rural population is positively associated with the area covered by forest. All the same, the non-significant effect of *urban population size* deviates from UPE's argument that urbanization raises consumption levels, which acts as a distal driver of deforestation (DeFries et al. 2010; Rudel 2013). The non-significant effect of urban population on forest cover (which does not change even after removing the variable for the *number of farms*) suggests that the effect of urban population at the local level is different than it is at the national level, even though I hypothesized that the mechanism is not contingent on scale. I discuss the implications of the positive estimate for rural population size in the conclusion.

Third, studies in spatial data analysis encourage researchers to explore whether or not the effect of an independent variable exhibits spatial heterogeneity, i.e., whether or not the effects of a predictor vary across space (Fotheringham 1999). Geographically weighted regression (GWR) (Fotheringham et al. 2002) is an exploratory tool that allows researchers to do this, and it has been employed by environmental social scientists who

[13] To address concerns of multicollinearity, I estimated variance inflation factors (VIFs) based on two cross-sectional OLS regression models (one for each time period, i.e., 2001 and 2006) and a first-difference, change-score OLS regression model with the same dependent variable and predictors. Because there are only two time periods, the results of the change-score model are identical to Model 1 (a panel model with two-way fixed-effects) (Allison 2009). For all three models, the maximum and mean VIF were less than 10, which is below the threshold conventionally used to identify problems of multicollinearity (O'Brien 2007). Thus, the non-significant effect of *urban population size*, in particular, is not due to an inflated standard error. Moreover, based on Ehrhardt-Martinez (1998), I also tested for the presence of an environmental Kuznets curve in urban population size. In this supplemental analysis, neither the log-linear nor the quadratic term for urban population was significant.

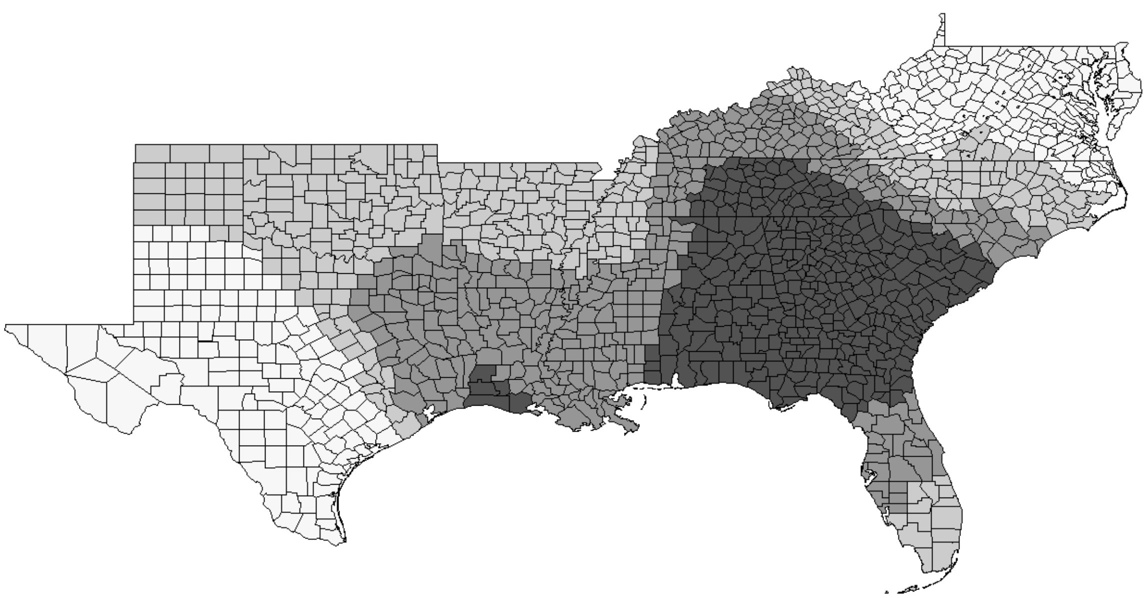
Table 2. Results from Panel Models and Spatial Panel Models with Two-Way Fixed Effects, 2001-2006 (N=2,846 county-years).

	Panel Model		Spatial Panel Model		Panel Model		Spatial Panel Model	
	Model 1		Model 2		Model 3		Model 4	
	b	SE	b	SE	b	SE	b	SE
Primary Variables								
Urban Population	-0.000	0.005	0.001	0.002	0.002	0.005	0.003	0.003
Rural Population	0.035 *	0.014	0.024 **	0.007	0.032	0.033	0.039 *	0.017
Rural Population x State (Texas is the reference)								
Alabama	. . .	. . .	. . .	. . .	0.071	0.147	0.007	0.074
Arkansas	. . .	. . .	. . .	. . .	-0.006	0.141	-0.030	0.067
Delaware	. . .	. . .	. . .	. . .	-0.181	0.856	-0.057	0.428
Florida	. . .	. . .	. . .	. . .	0.017	0.054	-0.025	0.029
Georgia	. . .	. . .	. . .	. . .	0.191 ***	0.057	0.079 **	0.028
Kentucky	. . .	. . .	. . .	. . .	0.072	0.104	0.034	0.053
Louisiana	. . .	. . .	. . .	. . .	0.400 ***	0.090	0.159 ***	0.045
Maryland	. . .	. . .	. . .	. . .	0.068	0.280	0.035	0.156
Mississippi	. . .	. . .	. . .	. . .	0.024	0.099	-0.014	0.049
North Carolina	. . .	. . .	. . .	. . .	-0.036	0.081	0.003	0.041
Oklahoma	. . .	. . .	. . .	. . .	0.105	0.117	0.038	0.055
South Carolina	. . .	. . .	. . .	. . .	0.184	0.145	0.148 †	0.077
Tennessee	. . .	. . .	. . .	. . .	0.037	0.123	0.019	0.061
Virginia	. . .	. . .	. . .	. . .	-0.045	0.038	-0.048 *	0.020
West Virginia	. . .	. . .	. . .	. . .	-0.029	0.149	-0.026	0.077
Controls								
Total Population	-0.037 *	0.018	-0.015 †	0.008	-0.068 **	0.020	-0.033 **	0.009
Number of Farms	0.002	0.004	0.001	0.002	0.002	0.004	0.000	0.002
Forestry Operations	0.001	0.003	0.001	0.002	0.000	0.003	0.000	0.002
Median Household Income	0.003	0.030	0.018	0.014	0.012	0.030	0.021	0.014
Total Payroll	-0.003	0.002	-0.003 **	0.001	-0.003	0.002	-0.003 **	0.001
Percent White	0.204 **	0.065	0.105 **	0.031	0.177 **	0.064	0.077 *	0.032
Percent Elderly	-0.041 †	0.023	-0.018 †	0.011	-0.039 †	0.023	-0.016	0.011
Percent Forest Area (2001)	0.005 ***	0.001	0.001 **	0.000	0.006 ***	0.001	0.001 ***	0.000
Year Dummy (2006)	0.017 ***	0.004	0.004 *	0.002	0.021 ***	0.004	0.006 **	0.002
ρ	. . .	. . .	0.820 ***	0.018	. . .	. . .	0.805 ***	0.019
λ	. . .	. . .	-0.607 ***	0.048	. . .	. . .	-0.594 ***	0.049
R^2 within	0.074		0.081		0.105		0.121	
Counties	1423		1423		1423		1423	
Years	2		2		2		2	

†p<0.1; *p<0.05; **p<0.01; ***p<0.001 (two-tailed tests) Note: Spatial panel models were estimated using a row-standardized, first-order queen contiguity weights matrix.

are examining whether or not the drivers of environmental change are spatially heterogeneous (e.g., Videras 2014). In this analysis, considering the significant slope estimate for rural population size in Model 2, I ask if its effect varies across space. To answer this question, I ran a GWR model using change-scores for the same dependent variable and predictors; I present the results as a map in Figure 5, which displays variation in the direction and magnitude of the *t-values* for the slope coefficient for *rural population size*. These results suggest that there is spatial heterogeneity in the effect of rural population size, with a strongly significant and positive effect clustered in and around Georgia. Based on the results from this exploratory tool, I further investigate the evidence for spatial heterogeneity, running a second spatial panel model with interaction terms between *rural population size* and *dummy-variables for states*, with *Texas* as the

Figure 5. Map of T-Values for Rural Population Change from Geographically Weighted Regression (Darker shading indicates higher t-values).



Note: The spatial weighting matrix used for the GWR model is an inverse-distance, adaptive bandwidth (e.g., Videras 2014; see also Fotheringham et al. 2002).

reference group. These results are presented in Model 4 and mostly support the findings from the GWR model, suggesting that the positive coefficient of *rural population size* is significant in and around *Georgia* but also in *Louisiana*, *South Carolina*, and *Texas* (the reference category). While the spatial panel model suggests that *rural population size* in *Virginia* is negatively associated with forest cover, the GWR model does not yield any significant negative slope estimates. (Note again that the conventional panel results in Model 2 show non-significant estimates for *Texas*, *South Carolina*, and *Virginia*.) Overall, the results from the GWR model and Model 4 suggest that the positive association between rural population size and forest cover is spatially heterogeneous, focused largely in and around *Georgia*. Nevertheless, considering that GWR is an exploratory tool, I present these results as motivation for future research to examine these issues in greater detail.

Lastly, I briefly report on the findings for the control variables. Based on the results from Model 2, only the estimates for *total payroll* and *percent white* are significant ($p<0.05$) and in the hypothesized direction. The negative coefficient for *total payroll* indicates that, as the size of the local economy grows, the amount of forest area declines, which is consistent with the treadmill of production theory (Schnaiberg 1980). The positive coefficient for *percent white* supports previous work on environmental inequality, which has found that whites live in areas with more tree cover and non-whites in areas with less tree cover (Harlan et al. 2008; Jesdale et al. 2013). Turning to the results in Model 4, the estimate for *total payroll* and *percent white* are still significant and the negative estimate for *total population size* becomes fully significant; the latter finding confirms research in STIRPAT and structural human ecology which emphasizes how overall population growth has a negative environmental impact (Dietz and Jorgenson 2013; Jorgenson and Clark 2011).

In summary, the conventional panel model and the spatial panel model yield different results; therefore, I broadly encourage researchers to consider issues of *both* temporal and spatial dependence in longitudinal research on the drivers of environmental change. Moreover, in terms of the effects of rural and urban population size on forest cover, the results from this local-level analysis present a picture that is different from the one usually seen in cross-national studies. In particular, *urban population size* has no effect on local forest cover whereas *rural population size* does, and this effect is in the direction opposite to what is hypothesized by EMT. That is, rural population growth at the county-level is associated afforestation. For local-level research, this finding also motivates a reevaluation of the way UPE has focused on urbanization as a distal driver of forest cover change. In the conclusion, I discuss these implications in greater detail and speculate on the reasons behind this unusual finding.

Conclusion

The above is an exploratory analysis to investigate the relative effects of rural and urban population size on forest cover at the local level in the Southern U.S. Even though I controlled for *both* temporal and spatial effects, I highlight the exploratory character of this chapter because previous sociological research on the systemic drivers of forest cover change mostly has been conducted at the national level. Nonetheless, as discussed above, the Southern U.S., compared to other Census regions, contains vast areas of forest and large rural and urban populations; it also has experienced high rates of change not only in terms of rural/urban population but also in terms of forest loss and forest gain. For these reasons, the Southern U.S. serves as an ideal context in which to carry out an exploratory analysis of rural/urban population as drivers of de/afforestation at the local level.

In the conclusion, I highlight two general results from this chapter: (a) the positive and (b) spatially heterogeneous slope estimate for rural population size. I find that rural population size is positively associated with forest cover change in many areas of the Southern U.S., even controlling for a variety of other factors, including variables not always operationalized in national level studies: the scale of primary production (i.e., the number of forestry and farming operations), the size of the local economy, residential affluence, age structure, racial composition, and the initial area of forest cover. Thus, controlling for several potentially confounding predictors, I find that rural population size still has an independent effect on forest cover, and this effect is uniquely positive. Further, I note that the association between rural population growth and afforestation in the Southern U.S. is spatially heterogeneous, and that this unique effect is most pronounced in and around Georgia.

On the one hand, the positive effect of rural population size and the non-significant effect of urbanization contrast with the claims made by EMT that rural living results in deforestation whereas city living contributes to afforestation. According to EMT, rural population growth is the reverse of modernization; it should *not* be associated with afforestation. On that note, the positive slope estimate for rural population size contradicts Mol's notion of permanency, or the assertion that ecological practices have become institutionalized in modern societies (2002: 94). According to Mol, a reflexive society would continue on a trajectory of environmental stewardship. Nevertheless, these findings suggest that the ecological modernization process *can* be reversed. That is, at least in terms of afforestation in the Southern U.S., rural living (*not* urban living) contributes to environmental stewardship.

On the other hand, the findings from this chapter are compatible with a particular dimension of the UPE framework. In term of forest resources at the local level in the Southern U.S., this chapter challenges the claim that cities are centers of growth that "require massive amounts of natural resources to sustain daily operations" (Jorgenson and Clark 2011: 240). To be clear, this finding pertains only to forest cover at the local level and does not contradict research that points to urbanization as a distal driver of deforestation at the national level. Nevertheless, I still encourage future cross-national research to incorporate other measures to test more explicitly the mechanism through which urbanization puts greater pressure on forest resources. For instance, this literature has argued that urbanization drives up domestic food production even though many low-income nations, which are included in cross-national studies, are net food importers (Ng and Aksoy 2008; Rakotoarisoa et al. 2011; Rudel 2013). So, at the national level, if the food consumed by urban residents is being produced in another country, then how exactly does urbanization contribute to domestic deforestation?

Meanwhile, at the local level, the positive association between rural population size and forest cover can be interpreted in two different ways, the first of which more directly implicates the UPE framework. First, rural areas in the Southern U.S. are characterized by a relatively slow socio-ecological metabolism. This argument is the converse to UPE's focus on the amplified urban metabolism, i.e., if urban areas raise consumption levels then rural areas depress them. The attenuating effect of rural population size is implied in growth machine and metabolic rift theories, which suggest that, *all else equal*, rural areas experience lower levels of human activity and present fewer opportunities to engage in commodified productive and consumptive activities. In environmental terms, rural living entails a slower socio-ecological metabolism and less natural resource use. In this chapter, however, there is no evidence that urban areas increase consumption of forest resources at the local level. For other environmental outcome (e.g., fossil fuel), previous research has argued that urbanization at the local level is multidimensional, and the different dimensions have countervailing environmental impacts (Elliott and Clement 2014). Given the non-significant effect of urban population size in the present chapter, future quantitative analyses of forest cover change might consider the different dimensions of urbanization (e.g., population size, density, and social organization) as separate independent variables. I begin to pursue that strategy in Chapters III-IV, examining the multiple dimensions of urbanization and growth and their differential effects on land use intensification.

Second, as I controlled for the initial extent of forest cover, the positive association between rural population size and forest cover could also mean the following: as the rural population declines so too does the area covered by forests. That is, many forests are located in rural areas with a variable number of residents; as the number of rural residents declines, i.e., as fewer humans inhabit these rural areas, the opportunity to exploit forest resources grows. Although the UPE framework is less directly implicated here, this second interpretation is still compatible with UPE, particularly its focus on the unequal economic exchange across the rural/urban divide (Lobao et al. 2007), in which rural areas are treated as supply depots and repositories for urban society (Burns et al.

2003; Buttel and Flinn 1977; Lichter and Brown 2011). But, in this case, it is not the size of the urban population that determines the degree of resource exploitation but the size of the rural population, which acts as a buffer to deforestation. This second interpretation does not necessarily imply anything about the consumptive and productive levels of rural or urban residents; it simply says that residents occupying rural land in certain areas of the Southern U.S. will slow down deforestation.

Which of these two potential interpretations can help explain the human dimensions of forest cover change in the Southern U.S. at the start of the 21st century? Additional research is needed to answer this question. Indeed, to distinguish between and to evaluate adequately these two possible interpretations requires a methodological approach that can address details about the historical, economic, and political contexts that have shaped the relationship between changes in rural/urban population and forest cover in specific areas of the Southern U.S. Again, there is strong evidence that the positive effect of rural population size varies across space, with the most robust findings in and around Georgia. Why is the positive association most pronounced in that area? Future qualitative analyses can examine this question in greater detail. Until then, the present exploratory analysis at the local level accomplished two objectives: 1) it evaluated competing hypotheses at the local level that have been tested mostly in cross-national research, and 2) to do this, it addressed analytic issues of temporal *and* spatial effects in quantitative research. On that note, while acknowledging the exploratory nature of this chapter, I hope future quantitative sociological work will consider both the substantive findings presented here (in terms of the positive and spatially heterogeneous effect of rural population size) and the benefits of a spatial panel model for longitudinal studies that examine the systemic drivers of environmental change.

CHAPTER III

INTENSIFYING THE COUNTRYSIDE: A SOCIOLOGICAL STUDY OF CROPLAND LOST TO THE BUILT ENVIRONMENT IN THE UNITED STATES

A revised version of this paper was published in *Social Forces* with my co-author Elizabeth Podowski, who helped collect the data for the dependent variable and write a paragraph describing the methodology she used to do the data collection.

Much quantitative research in sociology has examined the social drivers of environmental change. In this literature, population and economic factors stand out as primary drivers of a variety of environmental outcomes, including carbon emissions (Jorgenson and Clark 2012), deforestation (Austin 2010), ecological footprints (Rice 2007), and toxic emissions (Grant et al. 2002). Yet, aside from deforestation , quantitative sociologists have only occasionally paid attention to other land use changes (Axinn and Ghimire 2011; Befort et al. 1988). The present chapter contributes to the sociological literature on land use change (see also Entwisle and Stern 2005), looking at the systemic causes of cropland lost to the built environment in the United States. The conversion of cropland into roads, houses, commercial centers, and industrial facilities is a form of land use intensification, defined in the introduction as the drive of modern society to exploit progressively a particular parcel of land (Logan and Molotch 2007). While land use intensification may provide opportunities for development, it generally has a negative impact on the environment. Either through the conversion of undeveloped, open space to agricultural land or through the construction of the built environment, land use intensification generates pollution, degrades soil and water quality, and threatens biodiversity and biological productivity (Albrecht and Murdock 2002; Bradshaw and Muller 1998; Cronon 1992; Imhoff et al. 2004; Kaye et al. 2006; Lindenmayer et al. 2012; Mancus 2007; McDonald et al. 2008; Paul and Meyer 2001; Seto et al. 2012). Moreover, the environmental effects of land use change have been scaling up, directly and indirectly impacting the global biosphere and the planet's biogeochemical cycles, which is especially evident in the rapid rates of biodiversity loss, climate change and the disruption of nutrient cycles (Barnosky et al. 2012; Foley et al. 2005; Grimm et al. 2008; Rockström et al. 2009). In this light, land use intensification is a form of anthropogenic change with clear implications for environmental sustainability; the systemic causes of which should garner the attention of sociologists.

On that note, the present chapter examines the demographic and economic forces behind land use intensification. I focus specifically on the effects of population and economic growth on the transition from cropland to the built environment at the county-level in the United States between 2001-2006. The local land use change data for this chapter also come from the National Land Cover Database (NLCD). The NLCD is different from other national land use inventories, like the USDA's National Resources Inventory (NRI) (USDA 2009), because it also tracks permutations of specific land use transitions, making it possible to identify exactly how an individual parcel of land changed, if it changed at all. For the purpose of this chapter, the NLCD is used to quantify the total area of cropland that was lost specifically to the built environment between 2001-2006. More detail on the NLCD will be provided in the data and methods section. First, in this chapter, I review the relevant social and natural science literature on land use change and develop a theoretical perspective informed by demographic and political-economic approaches in environmental sociology (e.g., Axinn and Ghimire 2011; Bates 2009; Rudel et al. 2011; York et al. 2003). This framework provides a fresh set of hypotheses about how population and economic growth variably intensify land use at the local level in the United States. Second, the chapter discusses the data and methods that are used to test these hypotheses. With GLM models that can handle zero-inflated, nonnegative, continuous dependent variables, the area of cropland lost to the built environment is regressed on different measures of demographic and economic change. In the conclusion, the results of the analysis are discussed in terms of what land use intensification means for environmental sustainability and town-country relations (Bell and Korsching 2008; Buttel and Flinn 1977; Clement 2011).

Society, the Environment, and Land Use Intensification

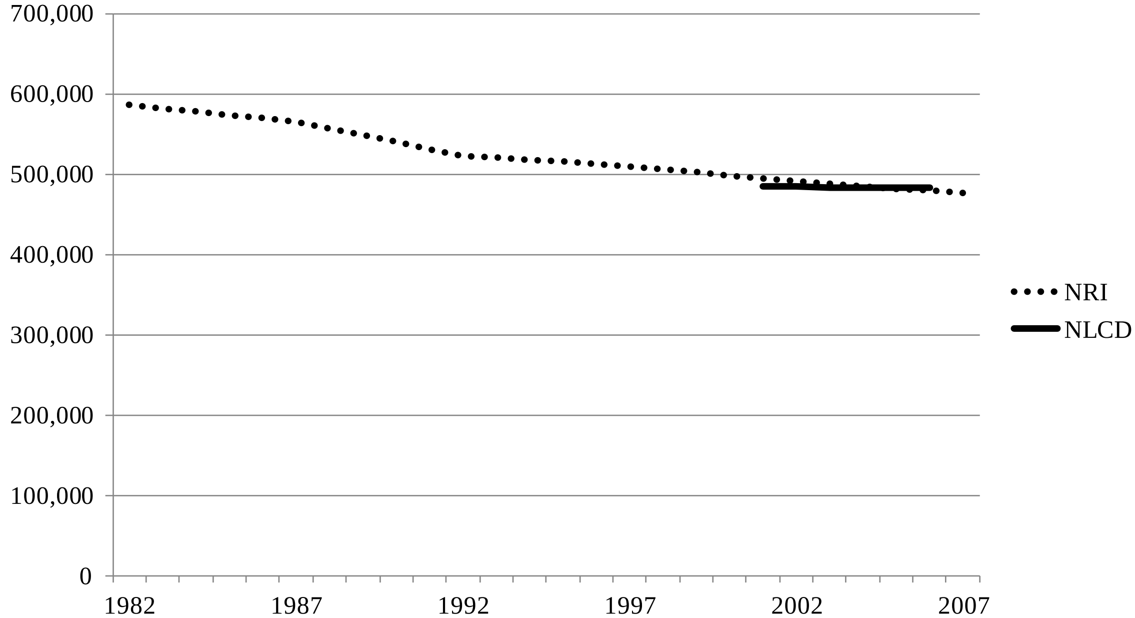
Sociologists from different perspectives have been studying land use change for over a century. Marx and Engels were concerned about how urbanization was disrupting the nutrient cycle (Foster 1999). The Chicago School sociologists Park and Burgess applied ecological metaphors, like succession, to study the changing landscape of cities (Young 2009). And, growth machine theory highlighted the conflicts over land use that emerge in a society where land is treated as a commodity with a use-value and exchange-value (Molotch 1976; Logan and Molotch 2007). While there are other sociological perspectives on the built environment (see Kilmartin 2002), growth machine theory puts the focus on land use intensification as a process by which socio-economic systems of growth strive to get the most value out of land as an exploitable natural resource. Especially in the United States, processes of growth drive the physical transformation of land from undeveloped space to agricultural land and the built environment through the construction of roads, houses, commercial centers, and industrial facilities. Urban sprawl is a commonly discussed form of land use intensification in which the construction of residential and commercial areas results in the relative decentralization of daily human activities, encroaching upon open-spaces like cropland (Anderson 1976; Bullard et al. 2000; Rudel 2009).

The sequence of land use intensification can be incremental, as implied in classical human ecology and central place theories (see Logan and Molotch 2007; Rudel 2009), or it can be abrupt and even discontinuous as highlighted in historical case studies (Cronon 1992). Whatever the pace of change, intensification is an ongoing process of environmental transformation, characterized by forward movement along a spectrum of land uses, bounded on one end by undeveloped wilderness and on the other end by the built environments of cities. A sociological perspective can shed light on the social drivers of landscape transformation at whatever stage in the intensification process. In this chapter, I examine the systemic causes of cropland lost to the built environment, a form of intensification that concerns two basic types of land use, each with distinct environmental impacts. In terms of cropland, much has been written about how industrial agricultural practices damage the environment, especially the water and soil cycles (Albrecht and Murdock 2002; Mancus 2007; Rockström et al. 2009). Nevertheless, when the built environment encroaches upon and covers up cropland, it reduces biological productivity, measured in term of net primary productivity (NPP) (Imhoff et al. 2004).

The covering up of cropland by the built environment also results in an increase in greenhouse gas emissions, directly through the release of the carbon sequestered in the plants and soils, and indirectly through the combustion of fossil fuel from the activities happening in, and needed to maintain, the built environment (Grimm et al. 2008). In fact, current estimates indicate that buildings contribute about one-third of all global greenhouse gas emissions (UNEP SBCI 2009).

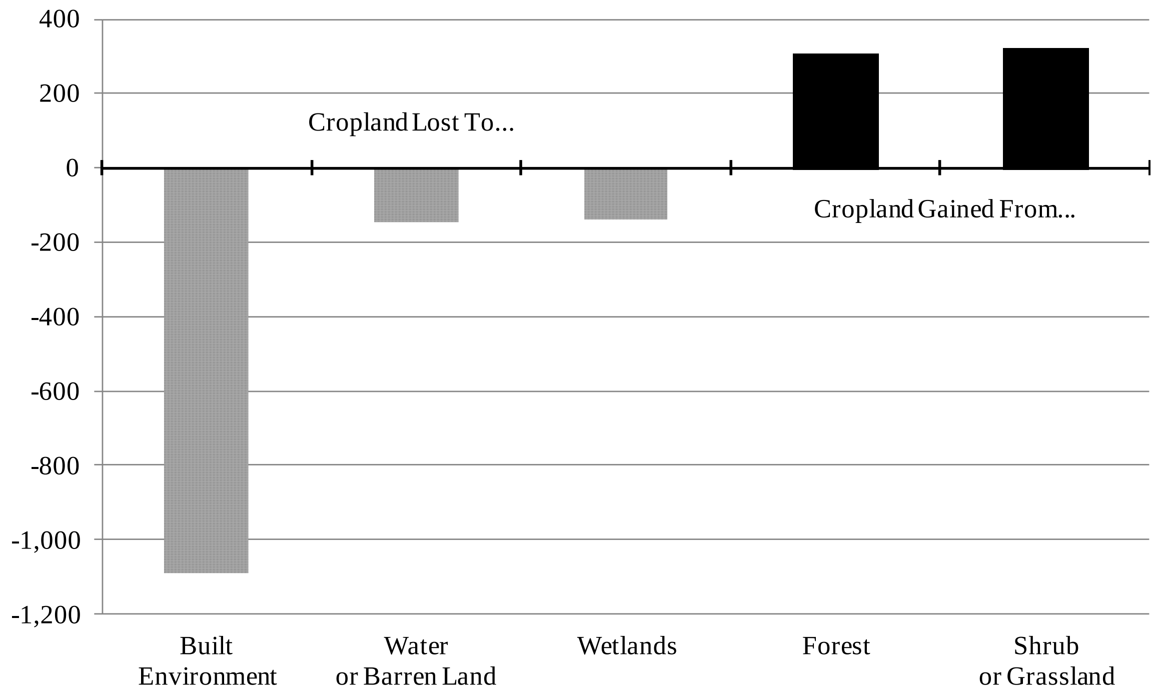
The USDA's National Resources Inventory (NRI) estimates that cultivated cropland in the United States declined roughly 18.7% between 1982 and 2007. Figure 6 displays the total area of cropland in square miles over time, as reported by both the NRI and the NLCD. Unlike the NLCD, the NRI does not identify specific permutations of land use transitions. According to data from the NLCD, cropland occupied about sixteen percent of total land area in the United States in 2001. Roughly 2,882 square miles of new cropland was *added* between 2001-2006, but about 3,628 square miles was *lost* during this time. Thus, there was a net reduction of approximately 746 square miles of cropland. This decline is shown in the slight downward slope in the NLCD line in Figure 6. Figure 7 is based on NLCD data and displays the area of cropland lost to and gained from five different general land cover types between 2001-2006. Of the 3,628 square of miles of cropland lost during this time, about 1,090 square miles were lost to the built environment, which represent a little more than 30% of all the cropland lost between 2001-2006. However, of the 2,882 square miles of cropland added, the built environment contributed only 0.5 square miles, or 0.017% of the total. Thus, without the losses to the built environment, there would have been a net increase of 343 square miles of cropland between 2001-2006. Moreover, the 1,090 square miles of cropland lost to the built

Figure 6. Total Area of Cultivated Cropland (in Square Miles) in the United States, 1982-2007



Note: The discrepancies between the two sources are the result of differences in data collection methodologies and land cover definitions. The NLCD will be described in the data and methods section of this chapter. For additional information on the NRI, see USDA (2009).

Figure 7. Net Change (Gain Minus Loss) in Area of Cropland (in Square Miles) by Land Cover Type, 2001-2006



environment corresponds to about a quarter of the nearly 4,400 square miles of built-up land added in the same five-year interval. In other words, a large portion of all new built-up land in the United States happened on cropland, and an even larger portion of the decline in cropland is attributed to the built environment. So, what is driving this form of intensification?

In sociology, previous national level research on land use has relied on aggregate measures of land cover change (e.g., national rates of deforestation) (Austin 2010; Ehrhardt-Martinez 1998; Rudel 1998), whereas prior studies at the local level have tended to examine changes within a single locality or region, making it difficult to generalize patterns and processes to an entire nation (Axinn and Ghimire 2011; Befort et al. 1988; Bradshaw and Muller 1998; DeFries et al. 2010; Perz et al. 2010; Rudel 2009). The latter focus has been due in part to the lack of consistent, nationwide local datasets on land cover change. The present chapter aims to fill this gap by using the NLCD, which allows us to ask if the social drivers of local land use change are generalizable beyond a local or regional case study to the larger population of localities. The question I ask is what systemic forces are at work in the intensification of the countryside *across the entire continental United States*. In this chapter, I frame population and economic growth as the primary social drivers of land use intensification, thereby placing this chapter in the broader sociological literature on the systemic causes of environmental change (Gould and Lewis 2008; York and Dunlap 2012). Yet, as mentioned, the quantitative literature on environmental change in sociology has not adequately examined changes in land use. Meanwhile, land is a natural resource, like fossil fuel and water, and, like other natural resources, is variably consumed as populations and economies grow and change. With the exception of deforestation (Austin 2010; Ehrhardt-Martinez 1998), quantitative sociologists have only recently started to examine land use change in this way, treating it as a resource that is variably consumed as populations and economies grow and change (Axinn and Ghimire 2011). To continue filling this gap, I draw on demographic and political-economic perspectives to discuss how different dimensions of growth uniquely intensify land use in the form of cropland lost to the built environment.

Population Growth

Sociologists and demographers have examined the environmental consequences of population change (Bates 2009; De Sherbinin et al. 2007; Pebley 1998), with quantitative researchers looking at the effects of population size, urbanization, and age-structure (Axinn and Ghimire 2011; York et al. 2003). With respect to natural resource use, research has shown that carbon emissions, for example, increase with population growth, urbanization, and a rise in the proportion of working age individuals (Jorgenson and Clark 2012; York et al. 2003). These factors have also been found to play important roles in studies of deforestation (e.g., Austin 2010; DeFries et al. 2010). I now briefly review the literature on the demographic driver of population growth in order to provide a framework that can illuminate how changes in population size variably affect land use intensification in the United States.

The positive effect of population size on natural resource use has often been framed as a neo-Malthusian argument, i.e., high fertility rates and low mortality rates result in population growth which depletes the planet's resources, posing a problem for sustainability as the number of babies increase and the number of deaths decrease (Bates

2009). Nevertheless, previous quantitative research on the systemic drivers of environmental change has not empirically disaggregated population growth into its two components: natural increase (births minus deaths) and net-migration (in-migrants minus out-migrants). This literature has used only a crude measure of overall change in population size (Jorgenson and Clark 2012; York et al. 2003); this is true even when the study's theoretical focus is on migration (Jorgenson and Burns 2007; see also Ehrhardt-Martinez 1998). Meanwhile, other research looking at the demographic drivers of environmental change (Axinn and Ghimire 2011; Bates 2009; Carr 2009; De Sherbinin et al. 2007; Geist and Lambin 2001; Pebley 1998) constitute a body of literature that suggests natural increase and net-migration should be treated as separate drivers of landscape transformation. In a meta-analysis of the social science research on deforestation, Geist and Lambin (2001) find that in-migration has been the primary demographic driver of tropical deforestation, with natural increase having a much smaller effect, relatively speaking. The primary effect of in-migration on tropical deforestation has to do with the potential for in-migrants to exploit significantly forest and other natural resources as they seek out economic opportunities (see also Jorgenson and Burns 2007). The environmental effect of in-migration is greater than the pressures resulting from the need to feed, clothe, and house new babies. Meanwhile, Axinn and Ghimire (2011) argue that new babies still drive up the demand for natural resources and that increasing the number of births specifically results in more public infrastructure and reduces the area of land under vegetation. However, I note that Axinn and Ghimire's study does not address the effect of net-migration and focuses only on the environmental consequences of births rather than natural increase (i.e., births minus deaths). And again, aside from the metaanalysis done by Geist and Lambin (2001), there have been no attempts by sociologists to disaggregate empirically population growth into natural increase and net-migration and systematically examine their differential effects on environmental change.

In the present chapter, I disentangle the relative effects of these two components of population growth on land use intensification, with the example of cropland lost to the built environment. Drawing from the literature cited above and the demographic dimensions of growth machine theory (Logan and Molotch 2007), I propose that net-migration has a bigger effect on land use intensification relative to natural increase. With net in-migration, as more people move into a locality than move out of it, the area will experience the construction of more roads, homes, shopping centers, office space, public infrastructure, and other impervious surfaces. Again, other sociological literature has theorized, but not empirically demonstrated, that net-migration is the primary mechanism through which population growth impacts the environment because it involves the movement of the most economically active individuals (Ehrhardt-Martinez 1998; Jorgenson and Burns 2007). As a result, net in-migration should increase the area of cropland lost to the built environment, thereby intensifying land use in the United States. Likewise, net out-migration curbs the construction of the built environment as a locality loses its most economically active individuals.

Compared to net migration, natural increase is expected to have a smaller positive effect on land use intensification. Population growth through natural increase entails a lag between an individual's birth and their transformation into prime actors in local production and consumption activities. Indeed, as Logan and Molotch (2007) point out, accommodating natural increase poses a challenge to local growth goals because of the time it takes for children to mature into active participants in the local economy (96). Still, based on the human ecological consequences of fertility and mortality in terms of resource consumption (Axinn and Ghimire 2011; de Sherbinin, et al. 2007), natural increase (i.e., more births than deaths) should intensify local land use. All else equal, natural increase translates into more public and private infrastructure, primarily in the form of residential construction and the expansion of schools and daycare facilities to accommodate more children. Likewise, natural decrease (i.e., more deaths than births) exerts the opposite pressure on the further development of cropland; it diminishes the drive to intensify land use. In communities where mortality outpaces fertility, there is a surplus of the built environment and cropland can remain as cropland, all else equal.

Economic Growth

Much previous sociological research examining the environmental consequences of economic growth has been conducted at the international level. These studies show that affluent nations have been shifting production to the developing world, thereby spatially displacing many of their environmental problems onto poorer countries through a process of ecologically unequal exchange (see Rudel et al. 2011). Indeed, not taking this spatial dynamic into account can result in what is known as the Netherlands fallacy, i.e., an affluent nation's environmental impact appears minor if one only looks within the country's borders because ecologically destructive economic activities (e.g., agriculture, manufacturing, and mining) are happening elsewhere (York and Rosa 2003). In quantitative studies on the drivers of environmental change, one way to address this international spatial effect is to include a variable for trade, foreign investment, or position in the global economy. Research on deforestation, for instance, has shown that increased trade in agricultural goods drives deforestation in Latin American and tropical countries (Austin 2010; DeFries et al. 2010). In these international studies, using trade or position in the global economy is important because the national measure of affluence (i.e., gross domestic product) cannot by itself adequately capture the environmental impact of economic growth, again because of the Netherlands fallacy (see also Jorgenson and Clark 2012).

In local level studies of environmental change, economic growth presents a comparable dilemma, which becomes resolvable through a synthesis of propositions from two theories: growth machine (Molotch 1976) and treadmill of production (Schnaiberg 1980). Reading these two theories together, one sees that environmental change at the local level depends not only the total level of economic production in an area but also on the affluence of the local residents living in the area. In fact, drawing on growth machine and treadmill of production theories, the level of economic production and the degree of residential affluence are presumed to have opposite effects on environmental change, and especially on land use intensification. On the one hand, the effect of economic production on land use change is akin to Schnaiberg's treadmill of production argument. Expanding production results in an increase in natural resource use, which in this case is the area of cropland converted to roads, houses, commercial centers, and industrial facilities. Thus, increasing the level of economic production is a positive driver of land use intensification, and it is comparable to the way economic growth at the international level results in more fossil fuel use, for example (Jorgenson and Clark 2012). Bigger economies consume more resources, either in the form of land or some other environmental outcome.

However, economic change at the local level is based not only on the level of production but also on the affluence of the local residents, which is presumed to have the opposite effect on land use intensification. In fact, residential affluence operates via Molotch's (1976) notion of aristocratic conservation, in which wealthy communities are able to treat their locality "as a setting for life and work, rather than as an exploitable resource" (328). As a result, those places that experience an increase in residential affluence can successfully organize to preserve open-spaces, like cropland, because they have the political-economic power to do so. In this way, aristocratic conservation puts the brakes on land use intensification and the construction of the built environment. Likewise, it should be noted that those places that experience a decline in residential affluence are less able to resist land use intensification, particularly because economic opportunities appear to be scarce. Declining affluence not only means more cropland lost to the built environment but is also generally connected to the introduction of environmentally destructive and toxic industries; these struggling communities are less able to influence decision making in order to enact and/or enforce policies that protect their health and the health of the environment (see also Freudenburg et al. 2009).

To summarize, the following hypotheses will be tested in the analysis using the land cover change data from NLCD:

H_1: Population growth from both natural increase and net in-migration will mean more cropland lost to the built environment

H_2: The effect of net in-migration on land use intensification will be greater than the effect of natural increase.

H_3: Increasing residential affluence will reduce the amount of cropland lost to the built environment.

H_4: Increasing the level of economic production will intensify land use.

Data and Methods

Dependent Variable

The dependent variable in this chapter is the area of cultivated cropland in square miles lost to the built environment between 2001-2006. Like the previous chapter, this measure comes from the National Land Cover Database (Fry et al. 2011). Thus far, mostly landscape ecologists have been incorporating these data into research projects (e.g., Theobald 2010), with some urban and population ecologists also taking interest (Reibel and Agrawal 2007; York et al. 2011). The current analysis pays attention to two of the land cover types identified by the NLCD: "cultivated crops" and "developed land." The NLCD defines cropland as "areas used for the production of annual crops, such as corn, soybeans, vegetables, tobacco, and cotton, and also perennial woody crops such as orchards and vineyards." Vegetation cover in these areas must be at least 20% cropland, and this land cover type includes all cropland actively being tilled. The NLCD defines developed land as "areas characterized by a high percentage (30% or greater) of constructed materials." This includes, for example, roads, single-family housing units, apartment complexes, city parks, golf courses, commercial spaces, office buildings, and factories. The dependent variable was quantified for all counties within the continental

United States using the Zonal Tabulate tool in ArcGIS and the "NLCD2006 From – To Change Index" file from the NLCD.

Independent Variables

There are four variables representing population and economic growth: *natural increase*, *net-migration*, *median household income*, and *total payroll*. Data for *natural increase*, *median household income*, and *total payroll come* from *USA Counties*. Data for *net-migration* come from the IRS's county-to-county migration data files. *Natural increase* is equal to the log of the number of births minus the log of the number of deaths in a county between 2001-2006. This turns out to be the log of the ratio of births to deaths. Similarly, *net-migration* is equal to the log of the number of in-migrants entering a county minus the log of the number of out-migrants leaving a county between 2001-2006, which is also the log of the ratio of in-migration to out-migration. *Median household income* represents residential affluence and is equal to the log of the county's median household income. *Total payroll* is a proxy for economic production and is equal to the log of the total payroll in a county, which includes all forms of compensation, such as salaries, wages, and officer and executive pay, among other items. *Median household income* captures the affluence of the residents of the county, regardless of whether or not they work outside of the county in which they live. *Total payroll* approximates the level of economic production within the county, regardless of whether or not the working individuals live inside or outside that county. Thus, whereas *median household income* represents affluence by place of residence, *total payroll* approximates economic production by place of work. For both measures of economic growth, 2006 constant dollars were calculated, logged, then first-differenced to get change scores for these two variables between 2001 and 2006.

Different controls are incorporated into the analysis. First, I include a measure of change in the population living in Census defined urban areas, defined as areas where at least 2,500 people live at a minimum density of 1,000 people per square mile. Environmental social scientists suggest that urbanization is an important factor to consider in studies of deforestation in developing nations (DeFries et al. 2010; de Sherbinin et al. 2007; Jorgenson and Burns 2007). Second, I control for the change in the percent of the population aged 65 years or older. Age is generally an important factor to consider in studies of land use change (de Sherbinin et al. 2007) and specifically relevant to an analysis of changes in farmland, where in the United States the average age of farm operators is about 57 years, and the fastest growing group of farmers is those aged 65 years or older (USDA 2007). Furthermore, older farmers are less likely to expect to sell their farms (Zollinger and Krannich 2002). Third, I include the following time-lagged variables for the year 2001: *population size*, *median household income*, *total payroll*, *urban population*, *percent aged 65 years or older*, *cropland area*, and *built environment area*. Because this is a panel model, these lagged variables control for ceiling and floor effects. For instance, there are several counties with no cropland to lose, and those counties with a lot of cropland have much to lose. It is also important to control for population size because counties with bigger populations will simply have more births and deaths as well as more in-migration and out-migration. Also, economic activity tends to concentrate in bigger populations. Finally, the Moran's I test for the dependent variable was significant ($I=0.210$, $p<0.01$), indicating significant spatial autocorrelation in the area of cropland lost to the built environment (see Figure 8). To address this, I compute spatial lags based on a queen, first-order contiguity matrix for the following three measures: *change in cropland area*, *cropland area in 2001*, and *built environment area in 2001*. Each spatial lag equals the average value of the independent variable for all neighboring counties (Anselin and Bera 1998) and controls for the potential spatial effects of being surrounded by counties that are losing cropland, have cropland to lose, and/or have land that has already been built-up.

Methods

To test the hypotheses, the dependent variable will be regressed on the independent variables in fixed-effects panel models, which controls for time-invariant factors (Allison 2009), such as whether or not the county has developable land, which can be influenced by both geographic and political characteristics (e.g., whether or not the county has steep mountains or has a history of land use controls) (e.g., Chi and Ho 2013). About 39% of all counties in the chapter lost no cropland to the built environment (1,212 out of N=3,108). The dependent variable is continuous, zero-inflated, with a long right-tail; in other words, it is a positively skewed, nonnegative dependent variable (see Figure 9). Modeling this distribution with OLS will create biased estimates (Long 1997). Some sociologists have used tobit models to analyze nonnegative dependent variables (e.g., Haynie and Osgood 2005; Jacobs and O'Brien 1998). More recently, other statisticians and social scientists argue that continuous, nonnegative, positively-skewed dependent variables should be modeled with a Poisson distribution using robust standard errors (Marwell and Gullickson 2013; Nichols 2010; Santos Silva and Tenreyro 2006;

Wooldridge 2010).[14] Poisson models treat zero values as real zeros and not as censored values of an underlying latent distribution, as the tobit model does. Santos Silva and Tenreyro (2006) find that Poisson produces more robust estimates than the tobit model. Therefore, the dependent variable is regressed on the independent variables using a Poisson generalized linear model (GLM) with a log-link and robust standard errors. As a test of sensitivity, slopes are also estimated with a negative binomial model and results juxtaposed with the estimates from the Poisson model. Even though Gould (2011) cautions against the use of negative binomial models because the model's assumptions generate biased coefficients when the dependent variable is measured continuously, simulations show that negative binomial can perform as well as Poisson. In analyses not reported here, I also estimate coefficients with zero-inflated Poisson and zero-inflated negative binomial models, using *area of cropland (2001)* as the predictor of excess zeros. Results are not meaningfully different.

Results and Discussion

The means, standard deviations and bivariate correlations for the dependent and growth variables are displayed in Table 3. The values of the univariate statistics for the dependent variable are not logged while the values for the independent variables are logged. The mean area of cropland lost to the built environment between 2001-2006 was roughly 0.35 square miles (the median was 0.02 square miles). The unlogged value for the mean *natural increase* is about 1.25, which is interpreted as 1.25 births for every

[14] Gould (2011) demonstrates the ability of Poisson with robust standard errors to estimate models with continuous dependent variables that are overdispersed.

Figure 8. Map of Area of Cropland Lost to the Built Environment, 2001-2006 (Darker shading indicates greater loss)

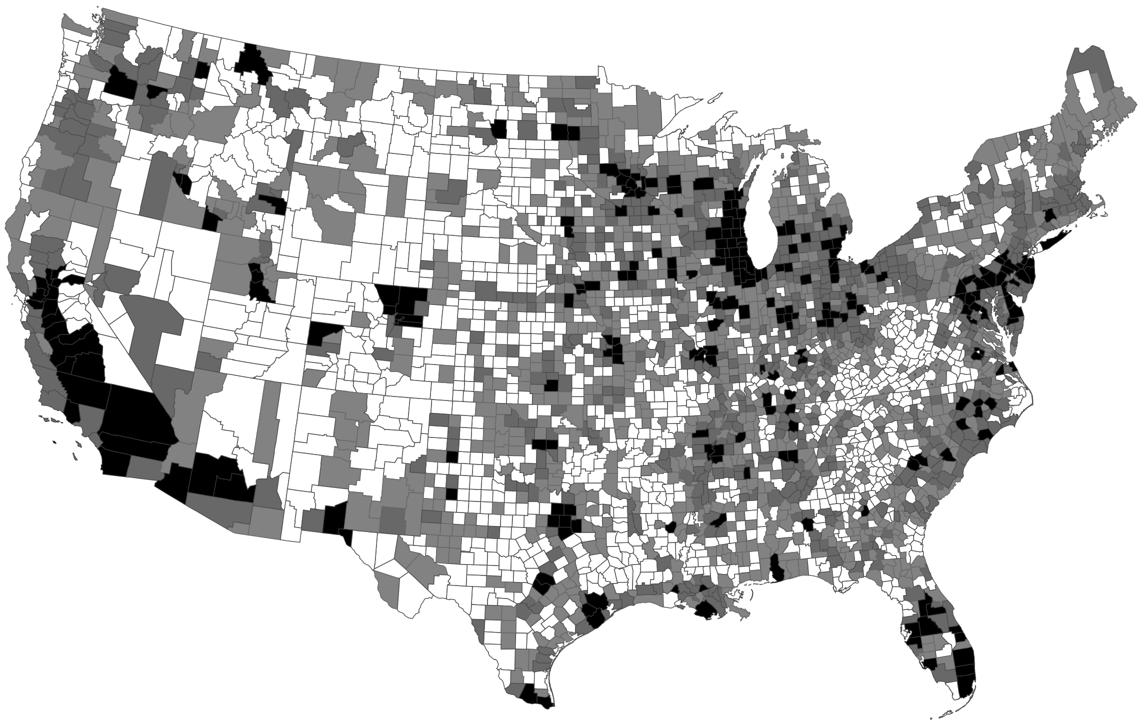
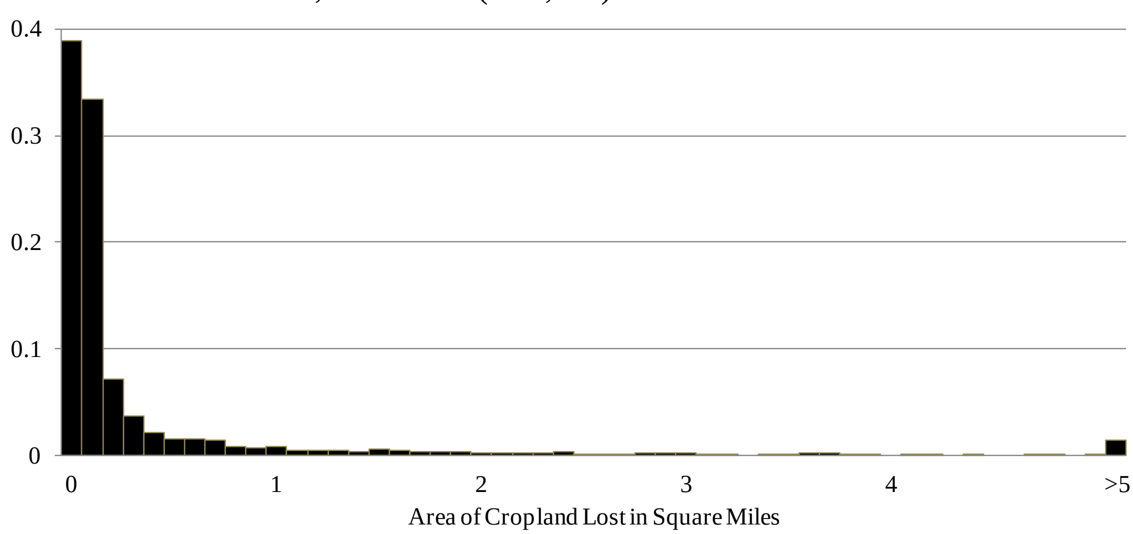


Figure 9. Distribution of Dependent Variable, Area of Cropland (in Square Miles) Lost to the Built Environment, 2001-2006 (N=3,108)



death. Thus, the typical county experienced more births than deaths. The unlogged value for the mean *net-migration* is about 0.96, which is interpreted as 0.96 in-migrants for every out-migrant. Thus, the typical county experienced net out-migration. Using the raw data from *USA Counties*, one sees that the mean *median household income* decreased by about $66.58 between 2001-2006 ($p<0.05$). Thus, while the typical county experienced a decline in residential affluence during this time period, mean total economic production increased by about 1.53%, as approximated by *total payroll*.

Table 4 displays the results from four regression models. Models 1-3 use Poisson GLM with log-link and robust standard errors to estimate slope coefficients, and Model 4 uses negative binomial GLM with robust standard errors. Since I use a log-link with Poisson, and negative binomial models are estimated with a log-link, the slope estimates for both types of models are interpreted as the percent change in the dependent variable for every one percent change in the independent variable, all else equal (Long 1997). (Note: All subsequent interpretations assume this all-else-equal condition.) Thus, with the log-transformation, all estimates are standardized (York et al. 2003), allowing us to compare the effects of two variables and conduct significance tests of the difference in magnitude between two coefficients. The results from the Poisson and negative binomial models for the growth variables are similar in terms of slope direction and significance levels but slightly different in terms of coefficient estimates. On balance, the effects of different measures of growth are only slightly sensitive to the form of regression analysis used to model the dependent variable (i.e., Poisson or negative binomial). All the same, considering Gould's (2011) caution against using negative binomial to model continuous dependent variables, I focus on the estimates from the Poisson models, and especially Model 3, for substantive conclusions.

Model 1 controls for neither the time-lagged nor spatially-lagged independent variables; Model 2 brings in the time-lags, and Models 3 and 4 includes all controls. Comparing the estimates from the first two models suggests the presence of ceiling and floor effects; the results change after controlling for the initial values of the independent and dependent variables measured in the year 2001. These ceiling/floor effects are especially noteworthy in the estimates for *natural increase* and *total payroll*. Including time-lagged controls attenuates the effects of the population variables and *median household income* and increases the estimate of *total payroll* in a positive direction. Most noteworthy, in Models 2 and 4, the estimate for *natural increase* is positive but not significant. Thus, in terms of population change, *net-migration* has a consistently positive and stronger effect on land use intensification. Looking at the estimate in Model 3, a one-percent change in *net-migration* results in roughly a 2.4 percent increase in cropland lost to the built environment ($p<0.001$).

Results from Wald tests of the difference in slope estimates between *natural increase* and *net-migration* indicate that the effect of *net-migration* is significantly bigger than *natural increase* in all four models, even when excluding controls for time-lagged and spatially-lagged independent variables in Model 1 ($\chi^2>42.15$; $p<0.001$). These results suggest that changes in migration levels represent a more potent demographic force behind land use intensification than changes in fertility and mortality, which challenges the neo-Malthusian focus on natural increase in theorizing the demographic

Table 3. Descriptive Statistics and Bivariate Correlations (N=3,108)

	Variable	Description	Source	Mean	SD	1.	2.	3.	4.	5.
Dependent Variable										
1.	Area of Cropland Lost to Built Environment	Square Miles of Cropland Lost to Constructed Materials and Impervious Surfaces (e.g., Roads, Single-Family Housing Units, Office Buildings, etc.) between 2001 and 2006, Not Logged	*NLCD*	0.351	1.522	1.000				
Growth Variables										
2.	Natural Increase	Ratio of Births to Deaths: ln(Number of Births between 2001 and 2006) - ln(Number of Deaths between 2001 and 2006) = ln(Births/Deaths)	*USA Counties*	0.221	0.407	0.266	1.000			
3.	Net-Migration	Ratio of In-Migrants to Out-Migrants: ln(Number of In-Migrants between 2001 and 2006) - ln(Number of Out-Migrants between 2001 and 2006) = ln(In-Migrants/Out-Migrants)	*IRS County-to-County Migration Data Files*	-0.037	0.171	0.182	0.231	1.000		
4.	Change in Median Household Income	Change in Median Household Income between 2001 and 2006. Values for 2001 and 2006 Adjusted for Inflation, Logged, then First-Differenced	*USA Counties*	-0.002	0.048	-0.045	-0.016	0.251	1.000	
5.	Change in Total Payroll	Change in Private, Non-Farm Payroll in a County between 2001 and 2006. Values for 2001 and 2006 Adjusted for Inflation, Logged, then First-Differenced	*USA Counties*	0.015	1.296	0.017	0.021	0.093	0.018	1.000

Table 4. Regression of Area of Cropland (in Square Miles) Lost to the Built Environment, 2001-2006 (N=3,108)

	Poisson (with log-link)						Negative Binomial (with log-link)	
	Model 1		Model 2		Model 3		Model 4	
	b	Robust SE	b	Robust SE	b	Robust SE	b	Robust SE
Population Growth								
Natural Increase	2.001***	0.131	0.130	0.197	0.170	0.203	0.221	0.174
Net-Migration	3.426***	0.509	2.248***	0.287	2.376***	0.259	2.647***	0.192
Economic Growth								
Change in Median Household Income	-5.398***	1.264	-2.517**	0.658	-2.043**	0.661	-1.169*	0.588
Change in Total Payroll	-0.023	0.024	0.357*	0.141	0.265*	0.117	0.379***	0.102
Controls								
Change in Urban Population	-1.363**	0.394	0.444	0.271	0.326	0.234	0.272†	0.163
Change in Percent 65 Years or Older	-5.650***	0.893	-3.076***	0.730	-3.019***	0.639	-1.839**	0.586
Time Lags								
Population Size (2001)	. . .	. . .	0.602***	0.151	0.787***	0.158	0.815***	0.128
Median Household Income (2001)	. . .	. . .	0.825***	0.202	0.822***	0.207	0.687***	0.175
Total Payroll (2001)	. . .	. . .	0.337**	0.098	0.226*	0.102	0.358***	0.091
Urban Population (2001)	. . .	. . .	0.296***	0.081	0.249***	0.066	0.175***	0.049
Percent 65 Years or Older (2001)	. . .	. . .	-0.475	0.316	-0.484	0.351	-0.376	0.265
Area of Cropland (2001)	. . .	. . .	0.703***	0.027	0.661***	0.049	0.708***	0.038
Area of Built Environment (2001)	. . .	. . .	-0.880***	0.116	-0.755***	0.109	-0.823***	0.103
Spatial Lags								
Change in Area of Cropland	. . .	. . .	. . .	. . .	-4.307**	1.508	-5.435***	0.947
Area of Cropland (2001)	. . .	. . .	. . .	. . .	0.070	0.052	0.079†	0.042
Area of Built Environment (2001)	. . .	. . .	. . .	. . .	-0.279**	0.098	-0.333***	0.069
Constant	-1.804***	0.068	-25.303	2.292	-24.157***	2.539	-25.056***	1.904
AIC	1.392		0.736		0.728		0.824	

† p<0.1, * p<0.05, ** p<0.01, *** p<0.001 (two-tailed significance tests)

drivers of environmental change. To be clear, population growth still emerges as a driver of environmental change, but the mechanism is more nuanced than the way environmental sociologists have previously theorized it (e.g., Austin 2010; Jorgenson and Burns 2007; York et al. 2003). These results strongly encourage researchers to reconsider how population growth is conceptualized in future sociological research on the systemic causes of environmental change and natural resource use; I incorporate this approach in Chapter IV.

The results for economic growth also present a complex picture about the ways in which economic production and residential affluence affect land use intensification. At the local level, these two dimensions of economic change have countervailing effects on cropland lost to the built environment. *Median household income* did not change much between 2001-2006; nevertheless, its slope estimate is still significant. As previously noted, the mean *median household income* actually went down by $66.58 during this time. For the nearly 52% of counties that experienced a decline in residential affluence, the negative slope estimate for *median household income* indicates an increase in cropland lost in the construction of the built environment. Likewise, an increase in *median household income* translates into less cropland lost to the built environment. Thus, increasing affluence by place of residence does put the brakes on land use intensification, which supports the notion of aristocratic conservation (Molotch 1976). Furthermore, expanding economic production by place of work, which has been approximated with the measure of *total payroll*, means more land use intensification. Looking at Model 3, for every one percent change in *total payroll*, there is roughly a 0.4% increase in cropland lost to the built environment. This result points to a treadmill of production in land use intensification; cropland, as a natural resource, is increasingly consumed as economic production by place of work grows.

Lastly, I briefly discuss the results for the control variables. When controlling for time-lagged independent variables in Model 2, the slope estimate for change in *urban population* switches signs and is no longer significant. These results suggest that residential density does not have an effect on the loss of cropland to the built environment, which diverges from previous literature emphasizing the effect of urbanization on deforestation (DeFries et al. 2010; de Sherbinin et al. 2007; Jorgenson and Burns 2007). Meanwhile, the slope for change in the *percent of the population aged 65 years or older* is consistently negative and even significant in Models 3 and 4. Thus, as a county's population becomes older, it is more likely to preserve cropland and spare open-space from the process of land use intensification, which is consistent with previous research on age and farm tenure discussed above (e.g., Zollinger and Krannich 2002).

Conclusion

This chapter took a sociological look at the intensification of the countryside in the form of cropland lost to the built environment. In the United States, about 1,090 square miles of cropland was lost to the built environment between 2001 and 2006; without this loss there would have been a net increase in cropland area. Drawing from the demographic and political-economic literature on the systemic causes of environmental change in sociology, the dependent variable was framed as an example of land use intensification and treated as an outcome of population and economic growth. The chapter makes two contributions to the sociological literature on the social drivers of environmental change. First, I used a consistent, nationwide dataset on local land cover change from the NLCD. This allowed us to test hypotheses about the social drivers of local landscape transformation and generalize the results to all localities across the continental United States. Second, I conceptualized population and economic growth in terms of natural increase, net-migration, residential affluence, and economic production. Results generally support the hypotheses. Compared to *natural increase*, *net-migration* has a significantly bigger effect on land use intensification, and the non-significant effect of *natural increase* (in Models 2- 4) challenges the traditional approach in environmental sociology which has focused on change in total population size. Looking at the results for economic growth, I found that increasing *median household income* is negatively related to land use intensification. This negative slope is explained by the growth machine process of aristocratic conservation, whereby changes in local affluence by place of residence is conceptualized as distinct from changes in economic production by place of work. Based on the notion of aristocratic conservation, increasing residential affluence means that local populations are better able to preserve the use-value of land and to resist growth projects that aim to enhance land's exchange value. In this situation, as Rudel (2009) has suggested, open space, like cropland, is saved, and land use intensification is slowed down or even halted. Likewise, declining residential affluence has the opposite effect, making communities less resistant to the forces that promote land use intensification projects. Drawing from related environmental sociological research in this area (e.g., Freudenburg et al. 2009; see also Rudel et al. 2011), I argue that residential affluence broadly structures the way communities relate to a variety of anthropogenic environmental changes, including land use intensification.

The intensification of the countryside through the loss of cropland to the built environment brings us to a discussion of environmental sustainability and town-country relations. The covering up of fertile cropland with the built environment is a socio-ecological expression of the town-country antithesis (Clement 2011; see also Bell and Korsching 2008; Buttel and Flinn 1977). Contrary to the claims of some scholars (e.g., Gans 2009; Harvey 2011; Wachsmuth 2012), I argue that the town-country divide is still relevant for studying the socio-ecological dimensions of land use. The construction of the built environment, which is a defining feature of cities today, is made possible through the uneven consumption of natural resources extracted from undeveloped and rural lands and often at the expense of open-space, like cropland. This uneven spatial-ecological dynamic has global environmental consequences; the built environment and city-life have direct and indirect impacts on carbon emissions, deforestation, nutrient accumulation, declines in net primary productivity, and agricultural production (Barnosky et al. 2012; DeFries et al. 2010; Foley et al. 2005; Grimm et al. 2008; Imhoff et al. 2004; Rockström et al. 2009). In this way, considering the forces of growth behind, and environmental consequences of, land use intensification highlighted in this study, I argue that modern processes of landscape transformation are contributing to society's ecological rift (Foster et al. 2010). Indeed, addressing these environmental impacts requires a sociological perspectives that makes connections between our contemporary socio-economic system of growth and land use change. To be sure, given our system of growth, it is not surprising that policy efforts intended to curb or slow down land use intensification are often met with limited or even no success (Logan and Molotch 2007; Logan and Zhou 1989; Robinson et al. 2005). On that note, as many natural scientists now argue, environmental sustainability, in which land use planning plays a central role, is only possible within the context of "socio-economic systems and institutions that are not dependent on continued material consumption growth" (The Royal Society 2012).

In conclusion, this chapter examined how demographic and economic growth are systemic drivers of environmental change, using the example of cropland lost to the built environment in the United States. As motivation for future scholarship, there are limitations of this analysis which, along with the results, pose four questions to be addressed by quantitative research on the drivers of environmental change in general and land use intensification in particular. First, does one see the differential effects of natural increase and net migration on other measures of environmental change? As a component of population growth, does net migration affect, for example, carbon emissions more than natural increase does? Answering this question will tell us to what extent environmental sociologists need to reconsider the mechanisms through which population growth impacts the environment and natural resource use. Second, this chapter distinguishes neither between the types of migration (e.g., amenity, housing, or labor) nor between the origin and destination of the migrants (i.e., domestic or international). Social scientists who research land use change have highlighted the value in examining the different dimensions of migration (Egan and Luloff 2000). Nevertheless, considering the availability of local migration data, making these distinctions in an analysis of local land use intensification *across the continental United States* will be difficult. Until better local level data for the entire nation become available, the results of this analysis suggest that net-migration, regardless of the type, origin, or destination, is a primary systemic force behind land use intensification. Third, other work in environmental demography points to the reciprocal effect of environmental change on migration (Entwisle and Stern 2005; Hunter 1998; Hunter 2005; Massey et al. 2010). Moreover, growth machine theory hypothesizes that there is a reciprocal effect of land use intensification on local processes of population and economic growth. While this chapter drew on the demographic and political-economic frameworks in environmental sociology to examine the systemic causes of land use intensification, a focus on growth machine theory could highlight the ways in which the construction of the built environment feeds back into these basic dimensions of local place making in the United States. Lastly, in this analysis, I used a two-period panel analysis to examine change over time; consequently, changes in the *pace* of landscape transformation, i.e., whether it is accelerating or decelerating, cannot be observed with these data from the NLCD. With more waves data, future studies can better evaluate if and how the pace of cropland lost to the built environment has evolved over time and space, and address Rudel's (2009) hypothesis that processes of landscape transformation have slowed down in late-modern America. Meanwhile, whatever the pace of land use change, the intensification of the countryside did continue into the early 21st century, being driven by local processes of population and economic growth. I look forward to future efforts to illuminate more fully the socio-ecological causes and consequences of this process.

CHAPTER IV

URBANIZATION AND LAND USE CHANGE: A HUMAN ECOLOGY OF DEFORESTATION ACROSS THE UNITED STATES

A revised version of this chapter is under review at *Sociological Inquiry* with Guangqing Chi and Hung Chak Ho as co-authors. My co-authors helped collect the data for an independent variable and write a paragraph describing this variable and the methodology they used to collect it.

Human ecology has had a contested yet long-lasting influence on sociological theory and research (e.g., Duncan 1961; Michelson 1968; Schnore 1961). This legacy is especially evident in the sub-discipline of environmental sociology where scholars emphasize demographic dynamics and the changing relationship between humans and ecosystems (Dunlap and Catton 1983; Freudenburg 2009; Freudenburg and Gramling 1989). On that note, in environmental sociology, concern about the natural environment has been explicit, rather than implicit or metaphorical as it was with the human ecology of the Chicago school (Wachsuth 2012; Young 2009; see also Michelson 1968). Furthermore, environmental sociologists have developed different strains of human ecology, one of which is called structural human ecology (Dietz and Jorgenson 2013; Knight 2009). Structural human ecology is a broad and flexible framework to study socio-environmental interactions; it is often employed in quantitative analyses to hypothesize and test the structural drivers of environmental change. Sociologists from this tradition have drawn from demographic and political economic theories to study how population change and economic growth affect a variety of environmental outcomes, including pollution, the consumption of natural resources, and changes in land use (e.g., Jorgenson et al. 2011; Jorgenson 2013; Lankao et al. 2009; Liddle and Lung 2010; Marquart-Pyatt 2013; Videras 2014; York et al. 2003; York and Rosa 2012).

Although early human ecologists had a distinct focus on urban processes (for overviews see Wachsmuth 2012; Young 2009), these urban issues have been given relatively little attention in the quantitative literature cited above. The present chapter seeks to fill that gap; I explicitly conceptualize urbanization as a structural demographic driver of environmental change. With a focus on urbanization, the following quantitative analysis builds on the human ecology literature in environmental sociology by emphasizing human ecology's urban roots. These roots constitute a line of inquiry which can offer a unique set of research questions about the environmental consequences of urbanization. I demonstrate the utility of this approach through the example of deforestation, a topic frequently studied by quantitative sociologists (e.g., Jorgenson and Burns 2007; Jorgenson et al. 2011; Shandra 2007).

I discuss the sociological and environmental research on deforestation in greater detail below. Here, I briefly acknowledge that deforestation is a form of anthropogenic environmental change, the tempo and drivers of which vary across time and space (Chew 2001; FAO 2012; Williams 2010). For example, during the past several decades, in the United States, forests around the country have been cut down in the construction of homes, roads, commercial centers, industrial facilities, among other impervious structures and surfaces (Foster 2010). The loss of forest cover to the built environment across the United States represents a more permanent, or "hard" (Foster 2010), form of deforestation that differs from the type of deforestation commonly examined in the quantitative literature (e.g., DeFries et al. 2010; Ehrhardt-Martinez et al. 2002; Jorgenson and Burns 2007; Rudel 2013).[15] In contrast to this cross-national research, I argue that hard deforestation, as a form of land use intensification, is a more direct environmental impact of urbanization, and I frame urbanization at the local level in terms of changes in the *size*, *density*, and *social organization* of a locality (Fischer 1976; Jacobs 1961; Molotch 1976; Wirth 1978). Drawing from human ecology, all three dimensions of urbanization are hypothesized to be structural yet proximate drivers of forest cover lost to the built environment.

To study how urbanization contributes to deforestation, I move the analysis down in scale to the local level where urbanization's multiple dimensions and their environmental impacts can be distinguished, theorized, and operationalized with US government data. The different measures of urbanization representing *size*, *density*, and *social organization* are joined with satellite imagery from the National Land Cover Database (Fry et al. 2011), which is used to quantify the dependent variable: the *area of forest cover lost to the built environment* at the county-level across the continental United States between 2001-2006. In this chapter, I first highlight some recent trends in deforestation across the United States; I then review previous literature on deforestation and urbanization, highlighting some data limitations confronted in cross-national research. Next, from human ecology, a set of hypotheses is formulated about the multiple dimensions of urbanization and their differential effects on hard deforestation at the local level. These hypotheses are then tested in two-way fixed-effects regression models. In the

[15] While "hard" deforestation signifies the loss of forest cover to the built environment, "soft" deforestation generally represents forest cover lost to agricultural practices and the timber industry.

conclusion, based on the results from the regression models, the concept of *urbanization's ecological dualism* is used to demonstrate that urbanization is a human ecological process with positive *and* negative environmental consequences. Moreover, I emphasize the implications of this concept for scholars and planners seeking to identify and ameliorate urbanization's multiple effects on the natural environment.

Deforestation across the United States

For millennia, humans have cut down and cleared forested lands to make space for agriculture and to exploit forest products for fuel and construction materials (Ponting 2007; Williams 2010). In the United States, agricultural practices and the European demand for lumber resulted in widespread deforestation, a process that was pushed westward as new land was settled between the 17th and late-19th centuries (Cronon 1992 and 2003; Foster 2010). This history was then followed by a long period of reforestation as farmland was abandoned and the lumber industry moved to other areas of the world. As a consequence, "soft" deforestation has increasingly become a problem in tropical nations, and since the 20th century, the scale and rate of forest loss has been unprecedented; deforestation now has a particular impact on biodiversity and the planet's carbon cycle, thereby contributing to the loss of species and global warming (FAO 2012).

While much deforestation around the world today is still the result of agricultural practices and trade (DeFries et al. 2010), as well as the exploitation of forests for fuel and timber (FAO 2012), the expansion of the built environment (i.e., roads, houses, commercial centers, industrial facilities, among other impervious surfaces) is making a noticeable impact in places such as the United States (Foster 2010). Based on the two waves of data available from the National Land Cover Database (NLCD), the area of the continental United States covered by forests declined from 787,055 square miles in 2001 to 775,418 square miles in 2006. Of the nearly 11,637 square miles lost during this time, about 1,265 square miles (or a little more than 10% of the decline) were lost to the built environment, an area roughly the size of Rhode Island. Based on the NLCD data, Figure 10 shows the area of forest cover lost to the built environment at the county-level between 2001-2006. The 1,265 square miles of hard deforestation represented a little more than 25% of the total 4,884 square miles of new built environment added across the United States between 2001-2006. Thus, the loss of forest cover is a notable consequence of the expansion of built environment.

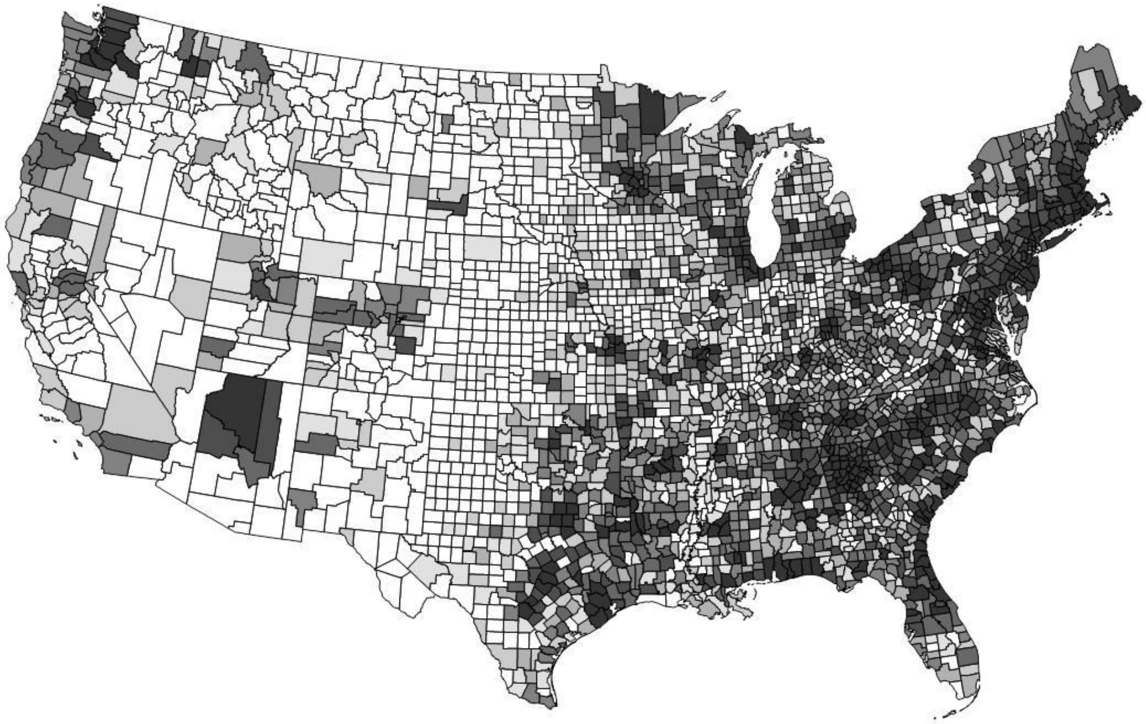
Many quantitative sociologists study the "soft" type of deforestation (i.e., cutting down trees to make space for farmland or to get access to forest products), which constitutes a greater area of land use change and, as already noted, results in substantial environmental impacts (e.g., FAO 2012). Here, I highlight briefly the ecological consequences of hard deforestation, which are distinct from the impacts of agriculture and the timber industry. Like the soft type of deforestation, hard deforestation directly entails the loss of habitats as forests are also cut down to make space for new development. However, in the latter type, as noted in the introduction, this loss is more permanent; once forested land is cleared for new development, it is then covered up with impervious structures and surfaces, thereby preventing reforestation, which is at least still possible in the event of "soft" deforestation (Rudel 2012b; see also Cramer and Hobbs 2007). Moreover, hard deforestation has direct and indirect impacts on ecosystem services and natural resource use by humans. Directly, by covering up the land, the built environment causes irreparable damage to biogeochemical processes like carbon sequestration, whereby the plants and soil are no longer able to sequester carbon dioxide from the atmosphere, obstructing an important ecosystem service. Indirectly, after constructing the built environment, society commits itself to a certain level of natural resource use required not only for the maintenance and repair of the built environment itself but also to fuel the various anthropogenic activities happening on this new impervious surface (Güneralp and Seto 2012). In summary, hard deforestation, as a form of land use intensification, has both direct and indirect, short-term and long-term environmental implications, making it distinct from the "soft" type of deforestation.

So, what are the structural drivers of this particular type of deforestation? Based on human ecology, how do the multiple dimensions of urbanization differentially affect the loss of forest cover to the built environment?

Urbanization and Human Ecology

Sociologists have examined the demographic, economic, historical, and political dimensions of changes in forest cover (e.g., Chew 2001). With limited exceptions (Macdonald and Rudel 2005; see also Bates 2006; Befort et al. 1988), most previous quantitative studies of deforestation by sociologists have been conducted at the national level, examining the demographic and economic factors associated with changes in forest cover cross-nationally (e.g., Ehrhardt-Martinez et al. 2002; Jorgenson and Burns 2007; Jorgenson et al. 2011; Rudel 2013; Shandra 2007). While the urban question figures more prominently in some of this research (e.g., DeFries et al. 2010; Ehrhardt-Martinez et al. 2002; Jorgenson and Burns 2007; Rudel 2013), the nation, as a level of analysis, precludes a more nuanced study of urbanization and its contribution to forest cover

Figure 10. Map of Area of Forest Cover Lost to Built Environment, 2001-2006 (Darker shading indicates greater loss.)



change. Here I address how the lack of adequate cross-national data complicates sociological analyses of urbanization as a structural driver of deforestation; this summary then provides the impetus for a human ecology of urbanization at the local level.

A the national level, operational measures for both forest cover change and urbanization are limited. First, while the satellite imagery used in more recent cross-national deforestation studies (e.g., DeFries et al. 2010; Rudel 2013) is an improvement over previous data on forest cover, these newer data still do not identify the area of forest cover lost exclusively to the built environment; they simply report measures of change in total forest cover (for discussion of the satellite imagery used in these studies see DeFries et al. 2010). Second, not only do definitions of urbanization vary by country (United

Nations 2005),[16] but also measures of population density are inadequate at the national level as they do not accurately reflect the density at which people actually settle at the local level (Liddle and Lung 2010). These data limitations have theoretical implications. For instance, using satellite imagery of forest cover change, Defries et al. (2010) and Rudel (2013) conclude that urban growth drives deforestation because city living increases the demand for natural resources and intensifies the exploitation of forest products (cf. Rees and Wackernagel 1996). Thus, the hypothesized mechanism through which urbanization is said to contribute to deforestation is indirect and remote. In these studies, urbanization drives deforestation because, the authors argue, urban lifestyles remotely put pressure on natural resource extraction in non-adjacent, distant lands, albeit within the country's boundaries. Meanwhile, due to the lack of adequate data, cross-national studies cannot test the idea that urban growth might have a direct and more instant effect on forest cover as land is cleared to make space for the built environment.

The analysis in this chapter seeks to address these issues. To that end, at the local level, I use consistently measured US government data on urbanization and advanced satellite imagery from the NLCD, which allows us to quantify the area of forest cover lost specifically to the built environment for the continental United States. Before I describe those data in greater detail, I first draw from human ecology to argue that urbanization is a multidimensional demographic process involving changes in population *size*, *density*, and *social organization* at the local level (Fischer 1976; Molotch 1976; Wirth 1978). Each of these dimensions are hypothesized to have independent and direct effects on changes in forest cover due to the expansion of the built environment. Before discussing

[16] While quantitative environmental sociologists using cross-national data recognize the limitation of the urban measure, these researchers must rely on the use of a statistical procedure (i.e., fixed effects) as a way to control for the variation in the country-specific definitions (e.g., Jorgenson et al. 2014).

these concepts and their hypothesized effects, I provide a brief overview of the human ecology framework to situate this chapter in the quantitative environmental sociology literature.

Human Ecology and Environmental Sociology

Human ecology has influenced two general lines of research in environmental sociology: 1) the environmental factors that affect human organization and sustenance activities and 2) the environmental consequences of demographic change (Buttel and Humphrey 2002: 37-44; Catton 1980; Rudel 2012b). Research in the structural human ecology tradition (Dietz and Jorgenson 2013) has generally (but not exclusively) followed the second tract, maintaining an emphasis on demographic dynamics. As mentioned in the introduction, structural human ecology frames these dynamics in terms of hypotheses about the structural forces of environmental change that mostly have been tested using quantitative techniques. This quantitative literature has conceptualized demographic forces as structural drivers of environmental change, and, with the exception of deforestation, this research has been conducted at multiple scales, not just at the national level (Jorgenson 2013; Lankao et al. 2009; Liddle and Lung 2010; Videras 2014; York and Rosa 2012). Following from this literature, the present chapter frames the multiple dimensions of urbanization as independent structural drivers of local environmental change, with the example of hard deforestation. Again, based on human ecology (e.g., Fischer 1976; Molotch 1976; Wirth 1978), these multiple dimensions are the *size*, *density*, and *social organization* of a locality. Each dimension and its hypothesized effect on deforestation are described below.

Population Size

In environmental sociology, much quantitative research treats population growth as an undifferentiated process of change in total population size (e.g., Jorgenson 2013; Marquart-Pyatt 2013; for an exception see York and Rosa 2012). While researchers recognize the endogenous and exogenous components of population growth (i.e., *natural increase* and *net-migration*), the operational emphasis remains on changes in overall size. Generally, population growth results in more natural resource use, even though the magnitude of the effect may vary over time and space (Jorgenson and Clark 2012). In terms of deforestation, however, the results for overall population growth are more mixed, with many social scientists arguing that migration is the primary component of demographic change that drives deforestation (Geist and Lambin 2001; see also Entwisle and Stern 2005). In quantitative cross-national studies of deforestation, as noted in the previous chapter, support for the effect of migration is purportedly found even though direct measures of migration are not incorporated into the analyses (e.g., Jorgenson and Burns 2007). For instance, Ehrhardt-Martinez et al. (2002) estimate rural-urban migration by subtracting rural population growth from urban population growth (234).

Following from the analysis in Chapter III, this chapter examines the independent effects of *natural increase* and *net-migration* on forest cover lost in the construction of the built environment, hypothesizing that these effects are positive. *Natural increase* (operationalized as births minus deaths) is endogenous growth (de Sherbinin et al. 2007), and in terms of deforestation, endogenous growth means more forests are cut down to make space for additional residential and commercial buildings as well as daycares and schools. All the same, *net-migration* (operationalized as in-migrants minus out-migrants) involves the movement of permanent residents into or out of an area and is expected to have a bigger effect on hard deforestation than *natural increase*. As more people move into a locality than move out of it additional roads and houses are built and other private and public infrastructure made to accommodate the net inflow of migrants. Residential migration typically involves the movement of the most economically active members of society, which further motivates the cutting down of forests for the construction of the built environment. Moreover, *net-migration* is more likely to increase the number of local households and thus housing units; whereas, when humans reproduce, *natural increase* is more likely to increase the density of existing households. Thus, the former adds to the built environment more immediately than the latter.

Population Density

As a dimension of urbanization, density cannot be adequately measured at the macro-level (Liddle and Lung 2010). Although national measures of density have been used in cross-national studies of deforestation (Ehrhardt-Martinez et al. 2002),[17] density is an indispensable measure of urbanization the local level. Human ecology has incorporated Durkheim’s insight on density into urban research (Schnore 1958), suggesting that higher population densities increase the frequency of interactions among people, which leads to occupational specialization and material interdependence, hence organic solidarity (see also Gibbs 2003). Catton (2002), an environmental sociologist, has pointed out how this strict Durkheimian approach suggests that density should reduce

[17] Ehrhardt-Martinez et al. (2002) incorporate a measure of density into a broader indicator of "population pressure" and thus do not examine the independent effect of density by itself.

demand for scarce natural resources. In line with this positive interpretation, other social scientists have drawn from human ecology research to highlight the potential environmental benefit of increasing population density in terms of improving the efficiency of natural resource use (Lankao et al. 2009; Liddle and Lung 2010). The beneficial effect of density has been widely accepted across the environmental social sciences, with many adding that higher densities also can minimize humanity's footprint on the land (Glaeser 2009; Newman 2006). In fact, while Schnaiberg (1980) acknowledged that density may contribute to environmental problems within cities (e.g., in the form of pollution), he clarified that "local environmental problems resulting from high density may resolve environmental pressures elsewhere" (75-6). That is, concentrating human settlements onto smaller areas of land creates the opportunity to preserve the environment outside of that densely-settled space. This idea calls to mind Glaeser's (2009) comment that "if you want to take good care of the environment, stay away from it and live in cities."

Based on these arguments, *population density* is framed as a demographic process that decelerates hard deforestation. Controlling for changes in total population size, rising *population density* is expected to slow down the loss of forests in the construction of the built environment. *All else equal*, as more people reside on the same amount of land, other areas can be spared from development. That is, denser forms of settlement ultimately preserve forested land (and perhaps open-space as well as cropland) from the forces of development. Likewise, across the United States, many localities experience decreasing density, or sprawling development. In this case, as there are fewer people living on the same amount of land (i.e., as the population spreads out), less land is spared from anthropogenic change. Ultimately, decreasing density means more forest is lost in the construction of the built environment.

Social Organization of a Locality

Catton (1980) distinguishes between the environmental consequences of population density and modern urban lifestyles (201-207), arguing that sociologists should consider not only the size and density of a settlement but also its social organization (see also Fischer 1976; Molotch 1976; Wirth 1938). Human ecologists in environmental sociology have argued that the transition from rural to urban living, and the accompanying change in social organization, alters humanity's socio-ecological metabolism and escalates natural resource use (Jorgenson 2013). Research on deforestation, discussed above and in Chapter II, has found that city life, as opposed to rural life, escalates the consumption of wood and agricultural products, which in turn remotely drives forest loss in non-adjacent lands (e.g., DeFries et al. 2010; Rudel 2013). The altered socio-ecological metabolism of urban areas also affects the direct and immediate connection between settlement change and hard deforestation. *Controlling for changes in population size and density*, the transition from rural to urban living entails a transformation of social relations, generating a unique set of institutions and activities which are absent in rural settings, such as shopping centers, hotels, restaurants, gyms, espresso bars, industrial facilities, and entertainment venues, including professional sports arenas as well as cultural spaces like art galleries, museums, and zoos. This transformation then escalates the consumption of forested land in the construction of the built environment. While environmental social scientists have argued that the social organization of urbanization and modern life results in indirect environmental impacts (Axinn et al. 2010; Defries et al. 2010; Rudel 2013), the effects of urban organization, in terms of changes in land cover, can also be direct. Over and above the area of forest cover lost to total population growth, a higher ratio of urban residents means more forests are cut down in order make space for the additional private and public infrastructure associated with the metabolic demands of city life.

Hypotheses

The following are three hypotheses being tested in the analysis.

H_1: Natural increase and net-migration will both have positive effects on hard deforestation, but the effect of net-migration on hard deforestation will be greater than the effect of natural increase.

H_2: Controlling for total population growth, increasing population density will slow down the loss of forests in the construction of the built environment.

H_3: Controlling for population size and density, the social organization of urban areas means more forest cover lost to the built environment.

Data and Methods

Dependent Variable

The dependent variables in this chapter is forest area lost to the built environment at the county-level across the continental United States between 2001-2006. Again, the data for the dependent variable come from the National Land Cover Database (Fry et al. 2011). As mentioned in the previous chapters, the NLCD identifies three types of forest cover: "deciduous", "evergreen", and "mixed forests" and four types of built-up land, or what the NLCD calls developed land; these are: "open-space", "low-intensity", "medium-intensity", and "high-intensity developed land". The values for the dependent variable are equal to the area (in square miles) of any of the three forest types lost to any of the four types of developed land. These values were aggregated to the county-level for all the 3,079 counties within the continental United States, again using the Zonal Tabulate tool ArcGIS and the "NLCD2006 From – To Change Index" file from the NLCD.[18]

Independent Variables

There are five variables representing the human ecological dimensions of urbanization: *natural increase*, *net-migration*, *population density*, *urban county*, and *percent urban population*. *Natural increase* and *net-migration* represent changes in population size, and *urban county* and *percent urban population* are two alternative measures of social organization. Data for natural increase and the percent urban population come from the US Census; net-migration data are from the IRS's county-to-county migration data files; population density is based on data from both the US Census and NLCD. *Natural increase* is equal to the change in population size due exclusively to the difference between the number of births minus the number of deaths in a county between 2001-2006. Similarly, *net-migration* is equal to the change in population size due exclusively to the difference between the number of in-migrants entering a county minus the number of out-migrants leaving a county between 2001-2006. *Population density* is equal to the number of people per square mile of land. *Urban county* is based

[18] In this chapter, I merge the independent cities in Virginia with the counties that immediately surround them, yielding a smaller sample than the previous chapter.

on the USDA's Rural-Urban Continuum Code; it is a dummy variable with a value of "1" indicating that the county has an urban population of at least 2,500 people. *Percent urban population* is the percent of the total population living in Census defined "urban areas". The Census defines urban areas at the tract-level as places with at least 2,500 people living at a density of 1,000 people per square mile (US Census Bureau 2011). For all urbanization measures, with the exception of the dummy-coded urban county, logged values were calculated for 2001 and 2006, and then these logged values were first differenced, yielding a proportional change-score.[19]

Several controls are incorporated into the analysis based on previous social science work looking at the systemic drivers of land use change (Clement and Podowski 2013; Krannich et al. 2011; Macdonald and Rudel 2005). These controls are: *median household income, percent white, percent elderly, percent bachelor's degree, residential building permits, total payroll, total employment, number of farms, number of forestry operations*, and dummy variables for the nine different *Census divisions* (with West North Central the reference category). Data for these variables come from the following sources: US Census, Census of Agriculture, and the USDA's Economic Research Service. For all control variables, logged values for 2001 and 2006 were calculated and then first-differenced, yielding proportional change-scores. To adjust for inflation, 2006 constant dollars were calculated for both median household income and total payroll before

[19] I note the operational differences between *population density* and *percent urban population*. Whereas the former measure makes no distinction between urban and rural areas, the latter does. In our study, the county with the lowest density in 2001 was McMullen, TX with nearly 64 people per square mile of developed land. In 2001, the county with the highest density was New York, NY (Manhattan) with 97,411 people per square mile of developed land. Nevertheless, according to the Census bureau criteria, *percent urban population* uses density as a threshold at the tract level for designating a population as urban versus rural. In this way, the counties of, for instance, Manhattan (97,411 people per sq mi), Washington DC (21,940 people per sq mi), and St. Louis (10,067 people per sq mi) are all equally 100% urban despite big differences in their densities.

logging and first-differencing. Lastly, I also incorporate controls for floor and ceiling effects, using $time_1$ (i.e., 2001) logged values of *population size*, *total land area*, *percent forest cover*, and *land developability index*. The *land developability index* is a measure of land availability and suitability for future conversion and development in a geographic entity. For each county, this index indicates what percentage of lands is available and suitable for development and conversion as of 2001. The land developability index was generated using spatial overlay methods based on data layers of surface water, wetland, federal/state-owned land, Indian reservation, built-up land, and steep slope, which are all seen as undevelopable. The methodology was described in Chi (2010). The data were obtained from the 2001 National Land Cover Data, the 2000 U.S. Geological Survey's Federal and Indian Lands, the 2000 Shuttle Radar Topography Mission's Digital Elevation Models of the National Aeronautics and Space Administration, and the 1996 Managed Areas Database of the University California-Santa Barbara's Remote Sensing Research Unit (see also Chi and Ho 2013).

Methods

To test the hypotheses, the dependent variable is regressed on the independent variables in change-score regression models. Because there are only two time periods, the results from a change-score model are equivalent to a two-way fixed-effects model, which controls for time-invariant variables with time-invariant effects, thereby minimizing omitted variable bias (Allison 2009). The dependent variable (i.e., forest cover lost to the built environment) is a non-negative, positively-skewed, continuous measure with several legitimate "0" values (i.e., many counties lost no forest cover to the built environment).[20] Consequently, the model is estimated with a Poisson generalized linear model with a log-link and robust standard errors (Gould 2011; Nichols 2010; Santos Silva and Tenreyro 2006; see also Clement and Podowski 2013; Marwell and Gullickson 2013).[21] Since the independent variables have been logged, and I model the dependent variable with a log-link, the use of change-scores yields an elasticity model, with the slope estimates representing the percent change in the dependent variable for every one-percent change in the predictor, holding the rest of the equation constant (see York et al. 2003). Lastly, because there is significant spatial autocorrelation in the dependent variable (*Moran's I* = 0.379;p<0.001), in a supplemental analysis not reported here, I include a spatially lagged dependent variable based on a first-order, queen-contiguity, row-standardized weights matrix. These results were not substantively different from the results discussed below.

Results and Discussion

Table 5 displays the univariate statistics for the dependent variable and primary independent variables. Across the continental United States, at the county-level, the mean area of forest cover lost to the built environment between 2001-2006 was 0.411 square miles. The largest areas of hard deforestation occurred around major metropolitan areas

[20] Based on the NLCD criteria, only five counties had no forest cover at all in either 2001 or 2006. These counties were Minidoka, ID; Cochran, TX; Crane, TX; Loving, TX: Winkler, TX. As mentioned above, to control for variation in the initial extent of forest cover and land developability, I include time$_1$ values for *total land area (2001)*, *percent forest cover (2001)*, and the *land developability index (2001)*.

[21] In addition to the Poisson generalized linear model described here, I incorporated the same variables into three other models, using negative binomial, zero-inflated Poisson, and zero-inflated negative binomial. For the zero-inflated models, I used the following as the inflation variables: *total land area (2001)*, *percent forest cover (2001)*, and the *land developability index (2001)*. In these supplemental models (results available upon request), the estimates for the primary urbanization variables are not substantively different from the results presented and discussed below.

in the South (e.g., Harris County, TX=29.5 square miles, Gwinnett County, GA=23.5 square miles, Fulton County, GA=21.3 square miles, Wake County, NC=19.5 square miles, and Mecklenburg, NC=16.9 square miles), while areas of little to no hard deforestation clustered in the Great Plains region. Based on the mean values for *natural increase* and *net-migration*, the typical county in the continental United States had slightly more births than deaths (1.24 births for every death) and lost more residents to out-migration than it gained from in-migration (0.96 in-migrants for every out-migrant). The average population density decreased by about 2 percent, with 31% of counties experiencing increasing density. Based on the USDA's Rural-Urban Continuum Code, nearly 79% of all counties have urban populations of at least 2,500 people. Lastly, there was a modest change in the mean value for percent urban population; the percent of the population living in Census-defined urban areas increased by about 0.6 percent between 2001-2006. In summary, between 2001-2006, the typical county in the United States experienced hard deforestation and modest urbanization, with more births than deaths, more in-migrants than out-migrants, lower population density, and a greater urban population.

The regression results are presented in Table 6. All models include controls for various measures of economic and demographic change in addition to controls for initial conditions at $time_1$ and dummy variables for the nine different *Census divisions* (with West North Central the reference category). For simplicity, I do not report the estimates for the Census division dummy variables. Again, all slope estimates are interpreted as

Table 5. Variable Descriptions and Univariate Statistics (N=3,079)

Variable	Description	Source	Mean	SD
Dependent Variable				
Forest Area Lost to Built Environment............	Area of Forest Cover (in Square Miles) Lost to Roads, Housing Units, Commercial Centers, Industrial Facilities, Among Other Human-Constructed Impervious Surfaces, 2001-2006	NLCD	0.411	1.488
Primary Variables				
Natural Increase..	Ratio of Births to Deaths, 2001-2006 (Logged, First Differenced)	US Census	0.216	0.407
Net-Migration...	Ratio of In-Migrants to Out-Migrants, 2001-2006 (Logged, First Differenced)	IRS County-to-County Migration Files	-0.037	0.171
Population Density	Population Per Square Mile of Developed Land, 2001-2006 (Logged, First-Differenced)	US Census	-0.024	0.063
Urban County...	Dummy-Indicator Based on Rural-Urban Continuum Code, 1=County Has an Urban Population of at least 2,500 People, 0=Otherwise	USDA	0.790	0.410
Percent Urban Population	Percent of Total Population Living in Census-Defined Urban Areas, 2001-2006 (Logged, First-Differenced)	NLCD	0.032	0.204

Table 6. Regression of Forest Cover Lost to the Built Environment on Size, Density, and Social Organization, 2001-2006 (N=3,079)

	Saturated (Model 1)		Population Size (Model 2)		Population Density (Model 3)		Social Organization - I (Model 4)		Social Organization - II (Model 5)	
	b	SE	b	SE	b	SE	b	SE	b	SE
Primary Variables										
Natural Increase......................................	0.246 *	0.122	0.379 **	0.122	. . .	. . .	. . .	. . .	. . .	. . .
Net-Migration..	2.158 ***	0.373	1.619 ***	0.307	. . .	. . .	. . .	. . .	. . .	. . .
Population Density	-5.457 ***	0.678	. . .	. . .	-4.409 ***	0.591	. . .	. . .	. . .	. . .
Urban County	0.600 ***	0.112	. . .	. . .	. . .	. . .	0.578 ***	0.116	. . .	. . .
Percent Urban Population........................	0.529 ***	0.108	. . .	. . .	. . .	. . .	. . .	. . .	0.320 **	0.117
Control Variables										
Median Household Income	-0.425	0.637	-1.019	0.695	-0.861	0.760	-1.332	0.769	-1.307	0.765
Percent White	-1.145	0.887	0.003	0.955	-1.419	0.935	-0.599	0.944	-0.735	0.935
Percent Elderly	0.577	0.584	-0.083	0.599	2.026 ***	0.459	1.595 **	0.459	1.578 **	0.458
Percent Bachelor's Degree.......................	0.142	0.575	0.559	0.511	1.289 *	0.558	1.298 *	0.538	1.256 *	0.523
Residential Building Permits...................	0.076	0.045	0.113 *	0.044	0.139 **	0.047	0.136 **	0.047	0.138 **	0.047
Total Payroll	0.034	0.039	0.026	0.043	0.039	0.041	0.027	0.043	0.038	0.042
Total Employment....................................	1.879 **	0.637	2.269 ***	0.601	3.777 ***	0.460	3.458 ***	0.464	3.417 ***	0.465
Number of Farms	0.391	0.283	0.332	0.277	0.409	0.259	0.339	0.267	0.355	0.266
Number of Forestry Operations	0.041	0.053	0.102	0.056	0.071	0.060	0.117	0.062	0.122 *	0.062
Population Size (2001)............................	1.111 ***	0.034	1.125 ***	0.034	1.137 ***	0.032	1.143 ***	0.034	1.166 ***	0.033
Total Land Area (2001)...........................	0.084	0.053	0.087	0.055	0.131 *	0.054	0.117 *	0.056	0.113 *	0.057
Percent Forest Cover (2001)	0.951 ***	0.050	1.012 ***	0.052	0.942 ***	0.051	0.984 ***	0.052	0.984 ***	0.053
Land Developability Index (2001)	0.189 **	0.066	0.201 **	0.067	0.318 ***	0.074	0.286 ***	0.076	0.319 ***	0.075
Constant...	-19.327 ***	0.579	-19.180 ***	0.624	-20.011 ***	0.600	-20.451 ***	0.632	-20.282 ***	0.640
BIC	-23986.020		-23943.980		-23962.550		-23924.340		-23922.100	
Max/Mean VIF	3.37/1.71									

*p<0.05; **p<0.01; ***p<0.001 (two-tailed significance tests)

Note: Models are estimated with a Poisson generalized linear model using a log link and robust standard errors; this is the appropriate model when examining a non-negative continuous dependent variable (Clement and Podowski 2013; Gould 2011; Marwell and Gullickson 2013; Nichols 2010). Slopes estimates for Census division controls (with West North Central as reference category) are not reported.

elasticities. The saturated model (Model 1) includes all measures of urbanization; Models 2-5 estimate the effect of each urbanization variable separately and are labeled as such in Table 6. Results are consistent across these two types of specification. Of the urbanization variables, *population density* had the biggest absolute effect, with a one percent increase in density corresponding to a 5.457 percent decrease in the area of forest cover lost to the built environment.[22] The other four measures of urbanization had the opposite effect; they intensified hard deforestation. *Net-migration* (i.e., exogenous growth) exerted the strongest pressure; a one-percent increase in the ratio of in-migrants to out-migrants resulted in a nearly 2 percent increase in the area of forest cover lost to the built environment. *Natural increase* (i.e., endogenous growth), *urban county*, and *percent urban population* also contributed to hard deforestation, albeit to a lesser extent. For instance, a one-percent increase in the percent of the population living in urban areas resulted in a 0.529 percent increase in forest cover lost to the built environment. Thus, while higher population densities helped to put the brakes on hard deforestation, variables representing the *size* and *social organization* helped to speed it up.

I now briefly report on the results from some of the control variables. The estimates for the following variables were positive and significant: *total employment*, *population size* (2001), *percent forest cover* (2001), and *land developability index* (2001). A one-percent increase in *total employment* resulted in a 1.879 percent increase in the loss of forest cover to the built environment; this is consistent with growth machine theory (Molotch 1976) and treadmill of production theory (Schnaiberg 1980). Controlling

[22] At the county level, between 2001-2006, the built environment did not experience any reforestation; thus, a negative slope estimate in these regression models indicates that the predictor had a suppressing effect on hard deforestation, i.e., higher densities slow down the loss of forest cover to the built environment.

for *total land area*, counties with more forest cover and more developable land in 2001 experienced higher rates of hard deforestation between 2001-2006.

In summary, the regression results provide broad support for the human ecology framework. In Models 1-2, *net-migration* has a significantly stronger impact on hard deforestation than *natural increase*. Thus, there was an asymmetry between the effects of the components of population growth, confirming previous research in environmental sociology (Clement and Podowski 2013) and arguing for environmental sociologists to disaggregate the exogenous and endogenous components of population growth. Indeed, the net-migration of residents involves the most economically active members of society, whose production and consumption behaviors intensify land use in a way that trigger hard deforestation. Similarly, looking at Models 4-5 (Social Organization I-II), I report that urban counties and growth in a locality's urban population both increased the anthropogenic pressure on forest cover in terms of hard deforestation. As seen in the saturated model, the impacts of these two measures were experienced over and above population size and density. Consequently, *regardless of changes in size and density*, urban counties and increasing urban populations magnified the impact on hard deforestation; i.e., the social organization of a locality had a direct and immediate effect on the loss of forest cover to the built environment. Lastly, the effect of *population density* was negative and significant in both Models 1 and 3; controlling for population size, this suggest that increasing density likely has a beneficial effect, helping to preserve forested lands from development projects.

Conclusion

This chapter built on the human ecology framework in environmental sociology (e.g., Catton 1980; Duncan 1961; Molotch 1976; Jorgenson 2013; York and Rosa 2012) to examine the link between urbanization and land use change with the example of deforestation. In the structural human ecology tradition (Dietz and Jorgenson 2013; Knight 2009), the present analysis makes a contribution to the quantitative literature by looking at urbanization as a structural demographic driver of "hard deforestation" across the continental United States. As a form of land use intensification, hard deforestation involves the permanent loss of forest cover to roads, houses, commercial centers, industrial facilities, among other human-constructed impervious surfaces. Cross-national studies of deforestation have not been able to examine adequately this type of environmental change due to a lack of sufficient data on urbanization and deforestation, as discussed above. To fill this gap, I conducted a local-level analysis using consistently measured US government data on urbanization and land cover from the NLCD. Thus, while cross-national studies emphasize an indirect link between urban lifestyles and forest loss in non-adjacent lands (DeFries et al. 2010; Rudel 2013), I argued at the local level that the link between urbanization and hard deforestation involves a different mechanism, in which the structural drivers are more directly and immediately connected to changes in forest cover.

In this chapter, following from the discussion in Chapter II, I conceptualized the multiple dimensions of urbanization in terms of the *size*, *density*, and *social organization* of a locality. Regression results support the hypotheses. In particular, *population density* helped slow down hard deforestation, but variables representing *size* and *social* *organization* had the opposite effect. Increasing the *size* of a locality (most notably through *net-migration*) triggered the loss of forests to the built environment, as did *urban county* and *percent urban population*, two alternative measures representing the *social organization* of a locality. The additional institutions and activities that are characteristic of urban areas amplify the socio-ecological metabolism of a locality, which means greater demands are placed on land development projects. These results suggest that urbanization is a multidimensional demographic process with countervailing impacts on the natural environment, which I describe as *urbanization's ecological dualism* (cf. Rees and Wackernagel 1996). At the center of this human ecological dualism is a demographic process that results in *both* positive and negative environmental impacts. Across the United States, increasing *density* helped spare forested land from the forces of development, while the other forms of urbanization had the opposite effect, pushing localities to cut down trees and cover the land with roads, houses, commercial centers, among other impervious structures. In this way, I argue that all three dimensions are structural and immediate drivers of hard deforestation.

While making the above contributions to the literature, I now highlight the present chapter's limitations as motivation for future research and policy. First, while I use the NLCD data to examine the loss of forest cover to the built environment, I do not distinguish between the different *types* of the built environment. Was forest cover lost to roads, residential development, shopping centers, or industrial facilities? In order to answer this question, researchers would need to conduct advanced GIS analysis of the NLCD data to identify the permutations of land use change more specifically. Second, based on the particular theoretical framework, the present chapter demonstrates the average effects of urbanization across the *entire* continental United States. Thus, even though I control for Census division as an independent variable, other researchers may be interested in asking whether and how the effects of urbanization might be spatially heterogeneous. Do the impacts of *size, density,* and *social organization* vary depending on the region of the country? Future studies can draw on other theoretical frameworks and different analytic techniques (e.g., geographically weighted regression; see Videras 2014) to answer these questions.

In terms of policy implications, urban planners and land developers across the United States and the world are acknowledging the need for more sustainable forms of settlement, as discussed in the introduction. Often called smart growth or compact development, the concept of density figures as a central topic in these discussions about sustainable urban planning (Nielsen 2014). The environmental performance of these projects appears promising, especially with respect to the relationship between density and fossil fuel use (Newman 2006); however, this performance has been subject to much scrutiny (e.g., Elliott and Clement 2014; Melia et al. 2011; Neuman 2005). Meanwhile, the above analysis also demonstrates the ecological benefit of density in terms of preserving forest cover from land development. Nevertheless, based on human ecology, I argue that urbanization must be represented by more than a singular measure of density; indeed, with their focus on density, planners and policymakers might fail to appreciate urbanization as a multidimensional process with countervailing environmental impacts. Clearly, a focus on density by itself offers a parsimonious approach to dealing with urban-environmental problems; relatively speaking, the costs of increasing density may be low compared to the environmental benefits gained through higher densities. Still, the sustainability of urban development will be limited without addressing urbanization's multiple dimensions. On that note, urban scholars and planners must find ways to tap into the environmental potential of urban density while also addressing the negative environmental consequences that stem from the heightened metabolic demands of urban lifestyles. I look forward to future efforts by researchers and planners who are working towards that more comprehensive goal of environmental sustainability.

CHAPTER V

CONCLUSION: SUMMARY OF FINDINGS, POLICY IMPLICATIONS AND EPISTEMOLOGICAL CONCERNS REGARDING URBANIZATION AND ENVIRONMENTAL CHANGE

In this conclusion, I first summarize the findings from the previous three empirical chapters. Second, I discuss the policy implications of these findings, emphasizing that urbanization entails more than just changes in population density. Lastly, I highlight two epistemological concerns to recognize the different theoretical and methodological approaches utilized by scholars studying the relationship between urbanization and environmental change.

In the three empirical chapters of the dissertation, I examine the ways in which urbanization differentially drives changes in overall forest cover as well as the loss of cropland and forest cover to the built environment. Results from regression analyses emphasize that the different dimensions of urbanization have countervailing impacts on landscape transformation. In Chapter II, I find that rural population size, not urban population size, is positively associated with overall changes in forest cover. This finding challenges ecological modernization theory (EMT). Proponents of EMT argue that modernization is characterized by an increasing urban population, which should put less pressure on forest resources, thereby helping to slow down deforestation (e.g., Ehrhardt-Martinez 1998). The findings from Chapter II dispute this claim, showing instead that rural population growth (which is the reverse of modernization) means less, not more, deforestation. The forest cover data utilized in that chapter are comparable to the data used in previous sociological studies on deforestation. Nevertheless, these data do not identify the area of forest cover specifically lost to the built environment. On that note, Chapters III-IV utilize a more nuanced dataset from the NLCD (Fry et al. 2011), which tracks the various permutations of land cover change. In these subsequent chapters, I use these data to focus on processes I broadly refer to as land use intensification, i.e., the loss of cropland and forest cover specifically to roads, houses, commercial centers, industrial facilities, among other impervious structures and surfaces. In these two chapters, instead of relying on a single measure of urbanization, I decompose urbanization into separate operational components representing the size, density, and social organization of a locality. As dimensions of urbanization, I find that net-migration and the percent of the population living in urban areas are positively associated with land use intensification while population density has the opposite effect. On the one hand, increasing the density of a locality helps to spare land from the forces of development. On the other hand, controlling for density, the social organization of urban areas entails additional institutions and activities, which are absent in rural areas and ultimately amplify the demand on land development projects. In summary, the results from the regression analyses of these three chapters demonstrate that urbanization is a multidimensional process that differentially transforms the American landscape.

I now turn to a brief discussion of the policy implications of these findings. As mentioned in the introductory chapter, much of the literature on urban sustainability has emphasized the importance of increasing the density of urban settlements (e.g., Newman 2006); urban planners and policymakers, with an interest in environmental issues, have been influenced by this research (e.g., Nielsen 2014). My project does not dispute the environmental benefits of density in terms of land use intensification; indeed, as more people reside on the same amount of land, other areas can be spared from development. To be clear, this finding justifies the emphasis that urban planners and policymakers have placed on denser forms of settlements in order to reduce the impact of cities. Nevertheless, population density is not the only way to measure urbanization; to talk about urbanization simply in terms of density precludes a more comprehensive approach to urban sustainability. To reiterate, based on the results of my project, I argue that urbanization has multiple dimensions, with both positive and negative environmental consequences with respect to landscape transformation. While a focus on density by itself offers a useful approach to dealing with urban-environmental problems, the sustainability of urban development will be limited if policymakers do not address urbanization's multiple dimensions.

Lastly, in terms of epistemological concerns, I first address the way social scientists frame the political-economic system of modern society and its relationship to urbanization. Second, given this framing, I then describe how urbanization is studied as an independent driver of environmental change. My intention here is not to provide an exhaustive investigation of these epistemological concerns but rather simply and briefly to acknowledge the alternative theoretical and methodological motivations behind socio-ecological research on urbanization and city-life.

First, in terms of the terminology that represents the political-economic context, my dissertation draws heavily from Logan and Molotch's (2007) growth machine theory. Thus, while I rely on the abstract term "growth", and present urbanization as a dimension of growth, I recognize that other urban scholars frame the political-economic system of modern society using the more specific term "capitalism" (e.g., Harvey 1982). Nevertheless, Logan and Molotch acknowledged that their work was deeply influenced by Marxian scholarship on cities, especially Harvey's (1982) *The Limits to Capital*. They initially framed their analysis of land use intensification as an effort to "steer a middle course" between a variety of approaches, including human ecology, social movement theory, Marxian literature, and many others. Logan and Molotch are clear about the focus of their argument: "People dreaming, planning, and organizing themselves to make money from property are the agents through which accumulation does its work at the level of the urban place" (12). In the penultimate chapter of their book, they argue that "corporate capital" (cf. Baran and Sweezy 1968) has had an increasing presence in the real estate industry and then cite Harvey's observation that "the distinction between capitalist and landlord has blurred concomitantly with the blurring of the distinction between land and capital and rent and profit" (quoted in Logan and Molotch 2007: 236-7). Thus, underlying their explicit use of the abstract term of growth is an implicit appreciation of the more concrete process of capital accumulation. I incorporate the same approach in my dissertation, abstaining from an explicit focus on capitalism when studying the relationship between urbanization and environmental change. Such a focus would call for a separate host of research questions and methodologies, which would be beyond the scope of my project. Nevertheless, the distinction between "growth" and "capitalism" has theoretical and practical implications for environmental social science and policy. This distinction is especially relevant when social scientists consider the way environmental problems are unequally created and distributed around the world. Global warming is a good example of this point. Historically, developed nations are disproportionately responsible the release of greenhouse gas emissions, so solutions to the problem should take this legacy into account. Thus, with respect to climate change, the developed and developing nations of the world face different questions requiring different answers. Briefly, the strategy should be focused on asking how to shrink resource use in the global North while "creating sustainable-egalitarian productive possibilities" in the global South (Foster 2011: 32). While that discussion is important to the study of modern environmental problems, it is beyond the scope of my dissertation project; I mention it here as an epistemological issue with the term "growth", which in the abstract is not sufficient for conceptualizing processes of uneven ecological exchange (Rudel et al. 2011). Meanwhile, as noted in the introductory chapter, I present my dissertation as groundwork for future research on urbanization and environmental change in terms of *alternative* political-economic systems (cf. Odum and Odum 2001), the pursuit of which will benefit from a socio-ecological analysis of urbanization in various political-economic contexts around the world. In this future research, I can ask whether or not the consequences of urbanization observed at the local level within the United States are also evident in other nations.

On that note, to reiterate, the bulk of my dissertation uses quantitative methods to examine urbanization abstractly as a driver of environmental change, which I operationalize as a separate variable (or variables) with environmental impacts that are theorized as independent from other demographic, economic, and political forces. Nevertheless, in the introductory chapter, in order to frame the dissertation as part of a broader literature on urbanization and environmental change, I discuss urbanization as part of a more complex process of socio-ecological change characterized by reciprocal relationships and feedback (i.e., urbanization both influences and is influenced by landscape transformation, natural resource use, and pollution). Additionally, in the introduction, I acknowledge that studies of urbanization and its environmental dimensions are contingent on the broader socio-historical context. Considering the relatively narrow focus of my dissertation project, the issues of feedback and socio-historical context represent a second epistemological concern. Indeed, a comprehensive socio-ecological analysis of urbanization and city-life would consider how they are not only part of this complex, reciprocal process of change but also how the particularities of urbanization and city-life are contingent on the broader socio-historical context. For instance, in developing the notion of a metabolic rift, Karl Marx argued, "Capitalist production collects the population together in great centres, and causes the urban population to achieve an ever-growing preponderance" (quoted in Foster 1999: 379). In this sense, broadly speaking, the unprecedented degree of urbanization in modern society is seen as a consequence of capitalism and its ability to concentrate human activities spatially. More recently, scholars have elaborated on the urban dimension of Marx's analysis, arguing that urbanization and the built environment represent an important secondary circuit for the absorption of capital and continued economic expansion (Burkett 1999: 119-25; Harvey 1982). Without going into greater detail, suffice it to say that this literature presents urbanization in a more complex light, as a process of change that happens *inter*dependently with (not independently of) other demographic, economic, and political forces. Again, my study treats urbanization as a multidimensional, independent driver of environmental change, thereby addressing only one direction of a more dynamic, reciprocal relationship. Still, this project constitutes new terrain in the discipline of sociology and thus represents important groundwork for developing a more comprehensive socio-ecological analysis that considers the social and environmental complexities of urbanization and city-life.

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