ESTIMATING THE VALUE OF ECOSYSTEM SERVICES PROVIDED BY THE RED HILLS REGION OF SOUTHWEST GEORGIA AND NORTH FLORIDA: A GIS APPROACH

by

EDUARDO M. RODRIGUEZ

(Under the Direction of Rebecca Moore)

ABSTRACT

The research presented in this paper attempts to value the ecosystem services provided by the Red Hills region of southwest Georgia and north Florida in a spatially explicit manner. Land cover transformations are projected using scenario analyses, and the resulting changes in total ecosystem service values are quantified. The aim of the study is to provide land owners and policymakers with better information about the non-market value of land, so that they can make better decisions regarding land use.

INDEX WORDS: Benefit Transfer, Ecosystem Services, Florida, Geographic Information Systems (GIS), Georgia, Land-use Change, Non-Market Valuation, Red-cockaded Woodpecker, Red Hills

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DEDICATION

For my parents, who have given me opportunity.

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CHAPTER 1

INTRODUCTION

1.1 BACKGROUND

Forests provide a full suite of services that are vital to human health and livelihood such as water filtration, carbon storage, wildlife habitat and diversity, scenic landscapes and recreational opportunities (USDA 2010). However, lacking a formal market in which they can be traded, these services are traditionally absent from society's balance sheet. The contributions of these services are often overlooked in public, corporate, and individual decision-making. Without a market, land owners have little incentive to consider their value when deciding the optimal use of their land. This leads to forests being undervalued, which makes them increasingly susceptible to development pressures and conversion. Land owners who only consider the timber value of land in forest production will be more likely to choose non-forest land use options, which provide more benefits to the land owner. It pays to convert land because the financial returns from conversion exceed those from conservation. This means fewer acres in forest production, reduced importance of the region in global forest markets, and loss of benefits to society from reduced flows of ecosystem services. Efficient land use decisions must consider the total economic value of each land use option, including market and non-market, use and non-use, values. When this total value of forested land, including the value associated with timber production and the other ecosystem services provided, is compared to the total economic value of alternative land uses, it is likely that more land would remain undeveloped ensuring sustainable flows of essential forest ecosystem services.

The Red Hills region of southwest Georgia and north Florida faces tremendous development pressure and it is unlikely that this pressure will be relieved anytime soon. Understanding the total economic value of undeveloped land in this region is therefore essential for making land use decisions that consider a broader array of long-term benefits to humans. While previous studies have attempted to estimate the worth of ecosystem services, none have focused specifically on the Red Hills region.

1.2 THE RED HILLS REGION

Different sources give conflicting information regarding the size of the Red Hills physiographic region. The Tall Timbers Research Station and Land Conservancy states that the region is a 300,000 acre (121,000 hectares) area located between Thomasville, Georgia and Tallahassee, Florida (Tall Timbers 2010). Meanwhile, Cox et al. (2006) defines the region as a broader area, bounded by the Ochlockonee and Aucilla rivers and the Cody escarpment. They estimate that it covers approximately 240,000 hectares, and encompasses four counties: Leon and Jefferson counties, Florida, and Grady and Thomas counties, Georgia. For this study, we define the Red Hills region as shown in Figure 1.1.

The Red Hills region is known especially for its scenic beauty and rich biological diversity. The plantations of the Red Hills contain a significant portion of the native longleaf pine forests remaining in the United States and the largest contiguous acreage on privately owned land. Once covering as many as 90 million acres from Virginia to Texas, less than 3 percent of the rapidly disappearing longleaf pine forests exist today. This extremely diverse system is home to imperiled animals such as gopher tortoises, red-cockaded woodpeckers and Bachman's sparrows. In fact, the Red Hills' population of red-cockaded woodpeckers, a federally endangered species that requires mature pine to nest, is the largest found on private lands in the Southeast (Tall Timbers 2010; The Nature Conservancy 2010).

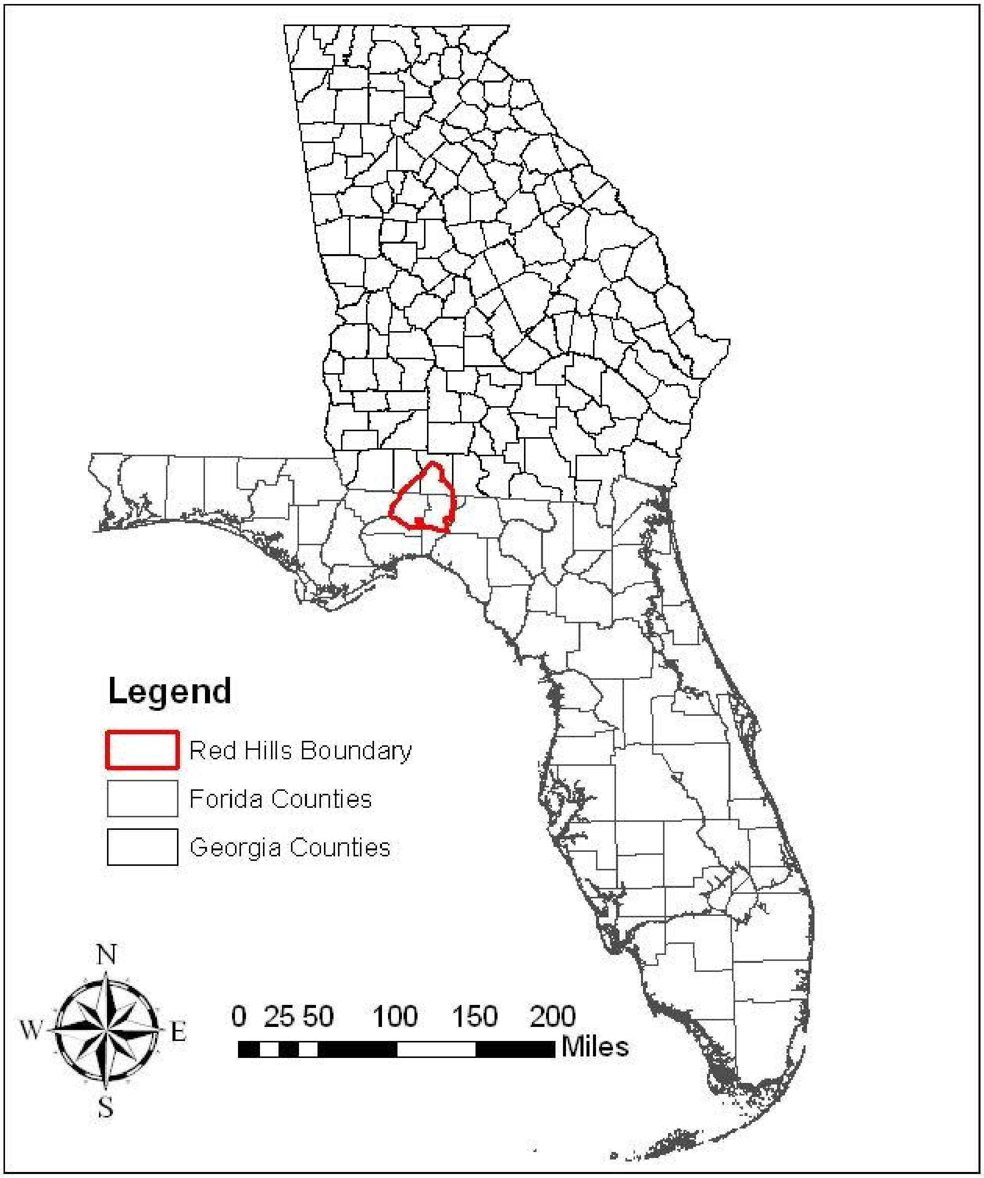

Figure 1.1: The Red Hills region

Urban sprawl is a phenomenon that, in the past few decades, has been fragmenting the Red Hills' landscape and putting some strain on its habitats. Tallahassee's sprawling growth over the years has consumed thousands of acres of forestlands in the Red Hills (Tall Timbers 2010). As of the 2000 census, there were approximately 150,000 people living in Tallahassee, up from 114,000 in 1980 and 125,000 in 1990. This trend is expected to continue, with estimates for the 2010 population surpassing 175,000. The percentage of Leon County, Florida's population living outside of Interstate-10 and Capital Circle has increased from 21 percent in 1970 to 49 percent in 2000 (US Census Bureau 2000). Similar sprawl is now occurring south of Thomasville. This rapid development is likely to have negative effects on the area's forest plantations which currently provide valuable ecosystem services and puts a strain on the fiscal resources of local government to provide the necessary infrastructure to serve that sprawl. This study will attempt to quantify and value some of these negative effects.

1.3 OBJECTIVES OF THE STUDY

The primary purpose of this research is to estimate the total value of ecosystem services provided by the Red Hills region. This region is ecologically valuable due to the many ecosystem services that it provides such as water filtration and supply, nutrient regulation, waste assimilation, soil retention, scenic beauty, disturbance prevention and habitat refugium (Tall Timbers 2010). As this study will show, forests and wetlands dominate the Red Hills region, and according to a study by Costanza et al. (1997a), these two land covers are among the most valuable when compared to other terrestrial land covers, in terms of services provided. They estimate, for example, that a hectare of forestland is 22 times more valuable than a hectare of cropland. Wetlands have an even higher value, 160 times the value of cropland.

A specific service provided by the Red Hills, and one that makes this region unique, is habitat provided for the endangered red-cockaded woodpecker. This species thrives in the Red Hills due to a large concentration of longleaf pine plantations in the area (Cox et al. 2006). Since the region is subject to strong development pressure from nearby urban areas, it is important to understand the full value of the land within the Red Hills boundaries in order to make well-informed land-use decisions.

This study applies a methodology for ecosystem services valuation across a landscape using Geographic Information Systems (GIS). With the increased use of GIS and the public availability of high quality land cover data sets, specific land cover categories such as forests, wetlands and cropland can now more easily be attributed with the ecosystem services that they deliver on the ground (Bateman et al. 1999; Eade and Moran 1996; Kreuter et al. 2001; Wilson et al. 2004). And with advances in benefit transfer techniques, values from the existing literature can more easily be adapted and transferred from one study site to another.

The specific goals of this research are therefore to:

1. Identify, describe, and quantify the ecosystem services provided by the Red Hills region.
2. Apply a spatially explicit value transfer methodology to estimate the total value of ecosystem services provided by the Red Hills region.
3. Conduct scenario analyses to evaluate the effect of urban and agricultural growth on the total value of ecosystem services provided by the Red Hills region.

1.4 ORGANIZATION OF THE STUDY

This thesis is comprised of five chapters. Chapter Two contains a review of the pertinent literature, including studies that have focused on ecosystem services, benefit transfer analysis, and spatially explicit valuation. In Chapter Three, the methodology for ecosystem service valuation is detailed. Chapter Four presents the results of this research. Chapter Five concludes the thesis with a summary and discussion of the findings.

CHAPTER 2

LITERATURE REVIEW

2.1 NON-MARKET VALUATION

Non-market valuation arose from the desire to include the natural environment in the decision-making calculus (Boyer and Polasky 2004). The value of goods and services that are sold in markets are represented through market prices. This allows for an easy cost-benefit analysis when making policy decisions that affect the value of marketed goods, since these market prices can be used to assess how much will be gained or lost from a specific policy proposal. However, those who advocate policies favoring the environment often find themselves at a disadvantage because they cannot express the gains or losses in values arising from environmental changes in monetary terms. Historically, environmental services have been grossly undervalued (Alig 1983). Sometimes, the lack of a monetary estimate of value for the natural world is treated as if the environment has zero value. This has lead to a lack of precise empirical data for measuring all environmental and wildlife values has resulted in superficial consideration of intangible and qualitative values, and greater emphasis on values measurable in monetary terms, even though these may be negligible in comparison (Van Dieren and Hummerlinck 1979; Kellert 1984). Researchers began noticing these problems decades ago, and valuation of the benefits of non-marketed resources and services in natural areas was identified as a major research need (Odum 1975 and 1977; Westman 1977; Alig 1983; Loomis and Hof 1985; Loomis and Walsh 1986). Since then, economists and others have attempted to supply monetary estimates of value created by the natural environment, as well as other things that are not bought and sold in a market but nonetheless have value (Boyer and Polasky 2004).

Many analysts object to the exercise of non-market valuation as applied to the environment (e.g., Sagoff 1988; Spash 2000). Some insist that valuation of ecosystems is either impossible or unwise, arguing that we cannot place a value on such intangibles as human life, environmental aesthetics, or long-term ecological benefits. However, Costanza et al. (1997b) argues that we already do so every day. He points out that when we set construction standards for highways, bridges, and the like, we value human life – acknowledged or not – because spending more money on construction would save lives.

Another argument against non-market valuation is that we should protect ecosystems for purely moral or aesthetic reasons, and that we do not need valuations of ecosystems for this purpose (Costanza 1997a). Advocates of this view see the pricing of the natural world as an example of the moral failings of the capitalist system in which everything is though of in terms of commodities and money. Yet Boyer and Polasky (2004) counter that the point of valuation is not to think in money or market terms, but to frame choices and make clear the tradeoffs between alternative outcomes.

A third argument against non-market valuation has more to do with practicality than with philosophical issues. Valuing ecosystems requires us to think about the value of the range of ecosystem "goods and services" produced by the ecosystem (Daily 1997). These goods and services must first be identified, then quantified, and finally valued in a common metric, ideally in monetary terms. Each of these steps presents practical challenges for trying to value ecosystems, which are complex, dynamic systems. Listing all of the services provided by an ecosystem may itself be a difficult task, and quantifying them is usually even more so.

Furthermore, while some ecosystem goods and services can be readily valued in monetary terms (e.g., commercially harvested timber), others present greater challenges (e.g., existence value of a species). However, a number of economic techniques have been developed to value these environmental goods in economic terms. These techniques are discussed in section 2.3.

Costanza et al.(1997b) in *An Introduction to Ecological Economics* counters the arguments against non-market valuation by insisting that while it is certainly a difficult task, we do not have the choice of whether or not to do it. He states that instead, the decisions we make as a society imply valuations. And that we can choose to make them explicit or not; we can use the best available science and understanding or not; we can acknowledge the huge uncertainties involved; but so long as we are choosing and making decisions, we are doing valuation.

2.2 ECOSYSTEM SERVICES

The evaluation of ecosystem services has been one of the popular issues in environmental and ecological economics (Costanza et al. 1997a; Daily et al. 2000), which aims to analyze and quantify the importance of ecosystems to human well-being in order to make better decisions regarding the sustainable use and management of these systems (Chen et al. 2009). Brown et al. (2007) refers to “ecosystem service” as the latest environmental buzzword, and points out that it appeals to ecologists, who have long recognized the many benefits derived from well-functioning systems; to resource economists, who attempt to measure the value of natural resources; and to a host of others, including land managers and policy makers, who see opportunities for a more efficient and effective provision of basic environmental services flows.

Gretchen Daily's (1997) Nature's Services, is widely regarded as a seminal work on the topic of ecosystem services (Heal 1999; Brown et al. 2007; Fisher and Turner 2007). When defining ecosystem services, Daily offers the following:

> Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain, and fulfill human life. They maintain biodiversity and the production of *ecosystem goods*, such as seafood, forage, timber, biomass, fuels, natural fiber, and many pharmaceuticals, industrial products, and their precursors…. In additions to the production of goods, ecosystem services are the actual life-support functions, such as cleansing, recycling, and renewal, and they confer many intangible aesthetic and cultural benefits as well.

It is important to note the distinction that is made between ecosystem services and ecosystem goods. This dichotomy is not expressed in the definition from the Millennium Ecosystem Assessment (2003), which simply states that "ecosystem services are the benefits people obtain from ecosystems." Ecosystem goods are generally tangible, material products that result from ecosystem processes, whereas ecosystem services are in most cases improvements in the condition or location of things of value (Brown et al. 2007). This difference is key in this study of the Red Hills Region, since it aims to estimate the value of ecosystem services and not ecosystem goods.

2.3 ECOSYSTEM SERVICE VALUATION

Various methods have been used to estimate the value of ecosystem services (Costanza et al. 1997a). Most researchers and authors divide ecosystem service valuation techniques into two main categories: revealed preference approaches and stated-preference approaches. Within each category there are various valuation methods that are employed to obtain monetary values for several types of ecosystem services. Figure 2.1 documents these techniques and illustrates how they are related, while Table 2.1 provides a brief description of each technique (Garrod and Willis 1998; Liu et al. 2010a).

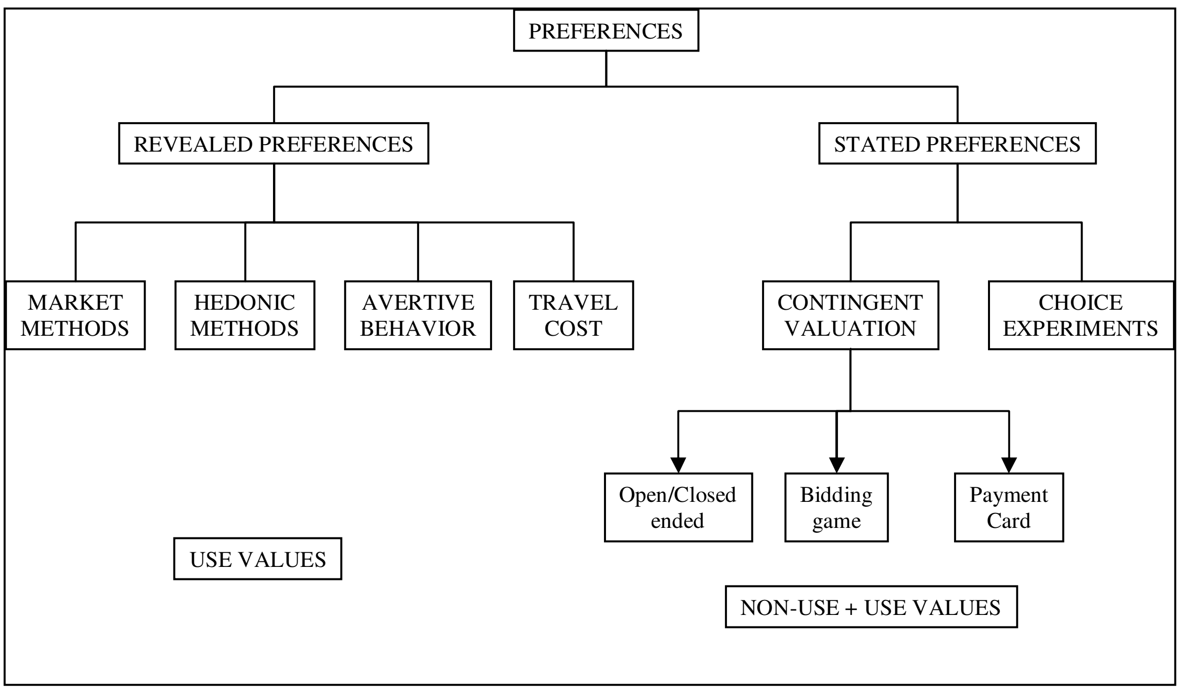

Figure 2.1 – Environmental valuation methods (Adapted from Garrod and Willis 1998)

The main distinction among monetary valuation methods is based on the data sources, that is, whether it derives from observations of human behavior in the real world (i.e., revealed-preference approaches) or from human responses to hypothetical questions (stated preference approaches). However, both of these approaches focus on individual choices and preferences, based on the fundamental assumption that individuals act so as to maximize their utility (Brown et al. 2007).

In the early stages of environmental forest valuation, the direct revealed preference methods, namely market methods, dominated. Then came the indirect revealed preference methods, such as travel cost (TC) and hedonic pricing (Markandya et al. 2002). In a later stage, the stated preference method of contingent valuation became predominant in order to include passive use value (Carson et al. 1999). This notion arose from the realization that a substantial portion of the total economic value of forest goods was not included by using the revealed preference methods (Stenger et al. 2008). Today, contingent valuation method is still the most widely used method for valuing ecosystem services (Vuletic et al. 2009).

Also worth mentioning is a method usually placed in a separate category. In contrast to the two methods discussed above, the *replacement cost* method does not rely on observing or modeling the behavior of individuals. Rather, this method computes the cost of replacing a lost environmental good or service, or conversely the replacement cost avoided if the environmental good or service is preserved. Economists are wary of cost-based approaches however, since estimates derived from them are not a based on preference, and therefore are not considered a measure of economic value (Liu et al. 2010a).

While all these valuation techniques are extremely useful for ecosystem valuation, the high costs of conducting original valuation research often lead researchers to employ another technique: the *benefit transfer* strategy, which is much more economical and much less time-consuming. This strategy is employed in this study therefore will be the focus of the next section.

A more thorough review of the literature on ecosystem service valuation using revealed and stated preference methods can be found in Adamowicz (1994) and Holmes and Boyle (2003).

Table 2.1: Ecosystem valuation methods (Adapted from Liu et al. 2010a)

Revealed Preferences
Market Methods: Valuations are directly obtained from what people are willing to pay for the services or goods (e.g., timber harvest).
Travel Cost: Valuations of site-based amenities are implied by the costs people incur to enjoy them (e.g., cleaner recreational lakes).
Hedonic Methods: The value of a service is implied by what people will be willing to pay for the services through purchases in related markets, such as housing markets (e.g., open-space amenities)
Production Approaches: Services values are assigned from the impacts of those services on economic outputs (e.g., increased shrimp yields from an increased area of wetlands.
Avertive Behavior: A service is valued on the basis of costs avoided, or of the extent to which it allows the avoidance of costly averting behaviors, including mitigation (e.g., clean water reduces costly incidents of diarrhea).

Stated Preferences
Contingent valuation: People are directly asked their willingness to pay or accept compensation for some change in ecological service (e.g., willingness to pay or accept compensation for cleaner air).
Choice Experiments: People are asked to choose or rank different service scenarios or ecological conditions that differ in the mix of those conditions (e.g., choosing between wetlands scenarios with differing levels of flood protection and fishery yields).

Benefit Transfer
The adaptation of existing ESV information or data to new policy contexts that have little or no data (e.g., ecosystem service values obtained by tourists viewing wildlife in one park used to estimate that from viewing wildlife in a different park).

The most commonly cited recent study that attempts to value global ecosystem services is that by Costanza et al. (1997a). The study presented a model for placing an economic value on different biomes and the services that they provided. Based on their models, the economic value of the world's ecosystems is $33 trillion per year, within the range of $16 trillion to $54 trillion. Since the entire economic output of the world is some $28 trillion, the world's ecosystems appear to be providing an annual flow of economic value 1.2 times that of the world GNP. Costanza et al. (1997a) goes on to state that because of uncertainties, these values should be considered a minimum.

Table 2.2: The global value of the world's ecosystems (*source:* Atkinson et al. 2001)

Biome		Economic value in 10^{12} \$(1994)
Marine:	Open ocean	8.4
	Coastal	12.6
Total marine		21
Terrestrial:	Forests	4.7
	Grass/rangeland	0.9
	Wetlands	4.9
	Lakes/rivers	1.7
	Cropland	0.1
Total terrestrial		12.3
Total		33.3

Atkinson et al. (2001) summarized Costanza's (1997a) economic values estimates by taking the central values only and condensing the categories of biome and ecological service (see Table 2.2). According to the results, coastal ecosystems provide around one-third of the economic value, with oceans, forests and wetlands also of major significance.

The estimates from Costanza et al. were widely criticized by economists for both theoretical and empirical reasons (Primm 1997; Masood and Garwin 1998; Pearce 1998; Toman 1998). Costanza et al. defends the exercise of valuing the services of natural capital 'at the margin,' which consists of determining the differences that relatively small changes in these services make to human welfare. However, Kreuter et al. (2001) points out that, because the last hectare of an ecosystem to disappear is likely to be worth much more than the first, simple multiplication of selected average values by all the units in the biosphere underestimated a potentially infinite social value of ecosystem services. In Roefie Hueting's New Scarcity and Economic Growth (Hueting 1980), he insists that we cannot measure the shadow prices of many non-market functions. In this context a shadow price is the willingness to pay for securing a change in a non-market value such as clean air, species preservation or scenic beauty. Hueting believes that the feasibility of estimating marginal willingness-to-pay (WTP) has always been close to zero, for many, but not all, classes of market environmental values. Atkinson et al. (2001) goes on to argue that the aggregated numbers from Costanza et al. (1997a) are not consistent with WTP for the simple reason that WTP is constrained by world income, $28 trillion. Ecosystem values could therefore not exceed that amount. Furthermore, Atkinson et al. (2001) refutes Costanza et al.'s values on the basis that they are not marginal values at all. They argue that if they were, they would relate to small changes in ecosystem services. Since the estimates from the study are clearly intended to be the value of the totality of the resources, Atkinson et al. (2001) concludes that there is no economic interpretation of virtually all the aggregated numbers in Costanza et. al (1997).

Despite all the criticism, Costanza et al.'s estimates of the value of ecosystem services represent the most comprehensive set of first-approximations available for quantifying the change in the value of services provided by a wide array of ecosystems. As a result, many studies have applied benefit transfer techniques using their estimates in order to value ecosystem services at a local scale (Kreuter et al. 2001; Williams et al. 2003; Liu et al. 2010).

2.4 BENEFITS TRANSFER

Benefits transfer, also referred to as value transfer (Desvousges et al. 1998), refers to the process by which a demand function or value, estimated for one environmental attribute or group of attributes at a site, is applied to assess the benefits produced by a similar attribute or site (Garrod and Willis 1999). This valuation method is generally advocated on the grounds of resource constraints. While there is growing sophistication of economic valuation methodologies, this growth is matched by the cost of conduction new studies for site-specific environmental change. Therefore, there is considerable interest in the cost-saving potential for generalizing values from one site to another when environmental conditions are suitably similar, since it is generally much less costly to engage in benefit transfer than to commission new research projects to analyze the benefits of every new project, policy, or regulatory change (Eade and Moran 1996). As such, value transfer has become an increasingly practical way to inform decisions when primary data collection is not feasible due to budget and time constraints, or when expected payoffs to original research is small (Environmental Protection Agency 2000), and it is now seen as an important tool for environmental policy makers.

There is, however, much debate concerning the validity of benefit transfer methods (Troy and Wilson 2006). One of the biggest potential pitfalls in value transfer occurs when values are drawn from study sites that are situated in very different contexts than targeted policy sites. It is therefore crucial for researchers to try and match the context of the source with the target as best as possible. Troy and Wilson (2006) anticipate that as the richness, extent, and detail of information about the context of value transfer increases, the accuracy of estimated results will improve. They argue that while value transfer is far from perfect, it is better than the status quo approach of assigning a value of zero to ecosystem services. Primary valuation research will always be considered the best strategy. However, value transfer represents a meaningful "second best" strategy and starting point for the evaluation of the environmental management and policy alternatives.

Table 2.3: The effect of land use change on ecosystem service values (*source:* Kreuter et al. 2001)

Land Cover Category	Change in cover 1976-1991 (ha)	Ecosystem service coefficient 1994 ($ ha^{-1} per year)	Change in ecosystem services 1976-1991 ($ per year)
Rangeland	-52,601	232	-14,187,931
Woodland	35,769	302	12,558,877
Bare soil	6694	92	715,996
Residential	5156	0	0
Commercial	9246	0	0
Transportation	-1891	0	0
Total	2373	-	-913,058
Avg. loss of value in ecosystem services ($ ha^{-1})			6

Costanza et al. (1997a) is itself an example of a study that employs benefit transfer. Its authors use point estimates from approximately 100 studies to derive the global average value of ecosystem services. Kreuter et al. (2001) in turn uses benefit transfer to apply coefficients published by Costanza et al. (1997a) to the San Antonio, Texas area, and to analyze how changes in land use affect the value of ecosystem services. In this study a sensitivity analysis is also conducted to determinate the effect of manipulating the coefficients on the estimated values. The distribution of land use changes in the study area is shown in Table 2.3. They report a 65%

decrease in the area of rangeland and a 29% increase in the area of urbanized land use between 1976 and 1991. Somewhat surprisingly, these land use changes were associated with only a 4% net decline in the estimated annual value of ecosystem services in the study area. The authors attributed the small decline to the neutralizing effect of the estimated 403% increase in the area of woodlands, which were assigned the highest ecosystem value coefficient. Nonetheless, the study highlights the appreciable losses in ecosystem service value that can accompany urbanization.

Table 2.4: Modifications to the basic Costanza type valuation for Scotland's ecosystem services value (*source:* Williams et al. 2003)

	Biome Valuation	
Ecosystem service	**Original Value (2001 $/ha)**	**Modified Value (2001 $/ha)**
Boreal forest		
Recreation	36	57
Culture	2	4
Grass rangeland		
Grass regulation	20.4	110.8
Soil formation	20.4	110.8
Tidal marsh/mangrove		
Disturbance regulation	1839	7337
Swamps/floodplains		
Water supply	7600	15095
Habitat/refugia	439	28
Recreation	491	575

Williams et al. (2003) also utilize Costanza et al.'s (1997a) global coefficients in an attempt to value Scotland's ecosystem services and natural capital. Some modifications were made to the coefficients so that they would be better suited for their study area. Williams et al. (2003) eliminated studies that were not appropriate for their region of study in order to obtain new coefficients. Values from studies with the most similar socio-economic conditions to

Scotland were thus employed. Table 2.4 summarizes the modification made to the estimates. While using the same methodology as Costanza et al. (1997a), the researchers derived an annual ecosystem services value of approximately 2001£17 billion ($\$24 \times 10^9$).

The advantages of benefit transfers have been recognized by a number of environmental agencies and in the mid nineties there was growing research interest in the development of "off-the-shelf' value libraries (Eade 1996). Today there are in fact several environmental valuation databases, such as the Environmental Valuation Reference Inventory (EVRI 2010), the EcoValue Project of the Gund Institute for Ecological Economics (2010), and the Natural Assets Information System, of Spatial Informatics Group, LLC (2010). These databases are being used more and more by researchers that employ benefit transfer methods. And the tendency, with the development of Geographic Information Systems, is for environmental valuation data to become increasingly spatial in nature.

2.5 VALUE TRANSFER USING GIS

In the past, Geographic Information Systems (GIS) have been widely used for environmental planning and monitoring, but were seldom used for environmental valuation. Through the years, technological advances in computer hardware and GIS software have encouraged a rapid growth in environmental GIS applications (Eade and Moran 1996). Yet while this growth has paralleled the development of economic theory and practice of environmental transfer itself, much less attention has been paid to the inherently spatial nature of many environmental values (Troy and Wilson 2006).

In their 1996 paper, Eade and Moran aimed to illustrate the importance and potential of using GIS to adopt a spatial approach to economic valuation of the environment. In particular, they wanted to demonstrate the utility for transferring site-specific benefit estimates. Using the Rio Bravo Conservation Area in north western Belize as a case study, they performed a benefits transfer exercise to give a spatial representation to the area's economic value. They achieved this by first mapping the "strength" or "quality" of the natural capital assets in the Rio Bravo, then using these maps to re-calibrate benefits estimates from alternative sites. The results of this process are "economic value maps," showing the benefit value of natural capital assets in two-dimensions.

Table 2.5: Ecosystem service values by land cover class and region ($/year in thousands).
***Source*: Ingraham and Foster (2008)**

Land Cover Class (acres)	Region (acres)						Total
	1 Pacific (3,383,793)	2 Southwest (2,413,706)	3 Midwest (592,449)	4 Southeast (2,930,74)	5 Northeast (298,294)	6 Mountain-Prairie (1,670,244)	
Open water (1,616,085)	$44,715	$41,078	$20,670	$309,024	$14,342	$37,625	$467,454
Forest (1,116,180)	$123,964	$71,441	$192,728	$330,507	$117,317	$108,031	$943,988
Shrubland (4,575,874)	$1,417,146	$958,823	$473	$2,094	$255	$139,944	$2,518,735
Grassland (1,385,381)	$12,603	$11,688	$400	$1,403	$0	$45,148	$71,242
Wetlands (2,595,706)	$1,176,292	$2,055,602	$1,668,501	$16,742,893	$619,927	$697,446	$22,960,661
Total (11,289,228)	$2,774,720	$3,138,632	$1,882,772	$17,385,921	$1,028,194	$1,028,194	$26,962,080

Ingraham and Foster (2008) used GIS in their study which attempted to value the ecosystem services provided by the U.S. National Wildlife Refuge System in the contiguous U.S. In order to obtain a total value, they determined the ecosystems present on the Refuge System in the contiguous 48 states, the proportion in which they are represented, and the dollar value of services provided by each. Land cover classes were used as an approximation of ecosystems present in the Refuge System. The land cover geospatial data was combined with a map of the Refuge System using GIS, and the acreage for each refuge and land cover class within the Refuge System was calculated. The researchers estimated the total value of ecosystem services provided by the Refuge System to be approximately $26.9 billion/year. Table 2.5 summarizes the values by land cover class and region.

Chen et al. (2009) used a GIS-based approach to map the value of ecosystem services at a much smaller scale: Tiantai County (1423.8 km^2) in Zheijian province of southeast China. In their study, selected components of natural products and tourism services in the study area were mapped as data layers in GIS, with each layer containing monetary values for every 25 m cell. The total direct use value of ecosystem services was estimated in RMB to be approximately 538 million Yuan in 2005 (8.2 Yuan = US$1), of which agricultural products, forest products and tourism products accounted for 65%, 30% and 5%, respectively. These results allowed them to identify critical areas in terms of resources protection and eco-environmental management purposes.

In their 2006 paper, Troy and Wilson present a decision framework designed for spatially explicit value transfer, and use it to estimate ecosystem service flow values and to map results for three case studies: Massachusetts, Maury Island, Washington, and three counties in California. They developed a unique topology of land cover for each case, and relevant economic valuation studies were queried in order to assign estimates of ecosystem service values to each category in the topology. The result was a set of unique standardized ecosystem service coefficients broken down by land cover class and services type for each case study site. Ecosystem values were summarized and mapped, and in the Maury Island case study, changes in ecosystem service value flows were estimated under two alternative development scenarios. For their Massachusetts case study, they estimated the total ecosystem service value flow for the entire state at $6.05 billion in 2001 dollars.

The authors followed a set decision rules for selecting empirical studies from the published literature that would allow them to estimate with sufficient accuracy the economic value of ecosystem services in their study area. They reviewed the best available economic literature, and selected valuation studies with the following characteristics: 1) those that were peer reviewed and published in recognized journals; 2) that focused on temperate regions in either North America or Europe and; 3) that focused primarily on non-consumptive use. The results from each study were then standardized to 2001 US dollar equivalents to provide a consistent basis for comparison. Once standardized, the resulting value estimates were assigned to the appropriate land cover categories (Wilson and Troy, 2003).

Table 2.6: Value of ecosystem services by land cover in Massachusetts. (*source:* Troy and Wilson 2006)

Land Cover Class	Ave. $/ha/yr (2001)	Lower Bound	Upper Bound
Cropland	$3,427	$3,427	$3,427
Forest	$2,430	$1,005	$4,934
Pasture	$3,412	$3,412	$3,412
Open Water	$2,427	$159	$7,374
Urban	$0	$0	$0
Urban Green	$8,471	$6,649	$10,293
Wetland	$38,167	$18,979	$78,476

The resulting coefficients from Troy and Wilson (2006) for each land cover class are presented in Table 2.6. A breakdown of each individual ecosystem services that is associated with each land cover class is provided in Wilson and Troy (2003), and presented in Table 2.7. Appendix B provides a full description of the ecosystem services used in their analysis.

Table 2.7: Individual ecosystem services by land cover class. (*source:* Wilson and Troy 2003)

Land Use Type	Ecosystem Services Used in Valuation
Wetland	Disturbance Prevention; Freshwater Regulation & Supply, Waste Assimilation, Aesthetic/Amenity, Soil Retention
Open Water	Freshwater Regulation and Supply, Habitat, Recreation, Aesthetic/Amenity
Forest	Climate and Atmosphere, Disturbance Prevention, Habitat Refugium, Recreation
Cropland	Aesthetic/Amenity, Soil Retention, Pollination
Pasture	Aesthetic/Amenity, Pollination
Urban Green Space	Waste Assimilation, Recreation

The most recently published study to perform statewide ecosystem service valuation using spatially explicit benefit transfer is that of Liu et al. (2010b). The authors estimate that the value of ecosystem services in the U.S. State of New Jersey is between $11.6 and $19.6 billion per year, depending on how inclusive they were in selecting the primary studies used to calculate ecosystem service values. They classified valuation studies into three categories according to their quality. Type A studies include peer-reviewed empirical analyses that used convention environmental economic techniques (e.g., Travel Cost, Hedonic Pricing and Contingent Valuation) to elicit individual consumer preferences for ecosystem services. Type B studies refer to technical reports, PhD theses, and government documents also using conventional environmental economic techniques. Type C studies are secondary studies summarizing primary valuation literature and they can possibly include both conventional environmental economic techniques and non-conventional techniques (e.g., energy analyses) to generate synthesis estimates of ecosystem services values. Table 3.4 summarizes their per acre estimates by land cover when including only type A studies, and when including all three types of studies. Estimates derived from Type A studies will henceforth be referred to as "Coefficients A" and those derived from Type A-C studies will be referred to as "Coefficients B."

Table 2.8: Value of ecosystem services by land cover in New Jersey. (*source:* Liu et al. 2010b)

Land Cover Class	Type A Ave. $/ac/yr (2004)	Type A-C Ave. $/ac/yr (2004)
Cropland	$23	$866
Forestland	$1,283	$1,476
Grassland/Herbaceous	$12	$77
Pasture	$12	$77
Open Water	$765	$765
Urban	$0	$0
Urban Green	$2,473	$2,473
Wetland	$8,695	$11,568

As GIS methods are increasingly adopted by economists working on non-market valuation research, the number of empirical examples of this type of work is rapidly growing. The papers described above are meant to provide an overview of the range of research currently available. The framework provided by Troy and Wilson (2006) served as guideline for decisions made in this research. Of particular use was their Massachusetts case study, as was the study by Liu et al. (2010b), since many of their ecosystem services coefficients are used as baseline estimates in this study of the Red Hills region.

CHAPTER 3

METHODOLOGY

The approach used in this study relies on a decision framework for mapping ecosystem service values presented by Troy and Wilson (2006). The framework was slightly modified for the purposes of this study, and consists of six core steps: 1) spatial designation of the study extent; 2) establishment of a land cover typology whose classes predict significant differences in the flow and value of ecosystem services; 3) meta-analysis of peer-reviewed valuation literature to link per hectare coefficients to available cover types; 4) mapping land cover and associated ecosystem services flows; 5) calculation of total ecosystem services values and breakdown by cover class and; 6) scenario analysis. These steps are described in the following subsections.

3.1 STUDY AREA DEFINITION

This step is often underappreciated, yet essential, since small changes in boundaries can have large impacts on ecosystem service value estimates. The Red Hills physiographic region does not have a formal political boundary, thus this study relied on general descriptions of its boundaries derived from the literature in order to create one in ArcGIS. The region has generally been described using the Aucilla river as its eastern boundary, the Ocklockonee river as its western boundary, and the Cody escarpment as its southern boundary (Cox et al., 2001).

Stream data (1:24,000-scale) obtained from the Georgia GIS Clearinghouse (GDT 1996), and a Digital Elevation Model of Florida (92 m cells) obtained from the United States Geological

Survey (USGS 1984) were used to digitize a boundary of the entire Red Hills Region (Figure 3.1). Within the DEM, an elevation cutoff of 20 meters was used to represent the Cody escarpment, which is consistent with descriptions of this natural boundary (Puri and Vernon 1964). One previous study by Cox et al. (2001) did in fact provide a map of the region. Though a digital copy of the shapefile used in that study was not available, the map was considered when digitizing the final boundary for this research.

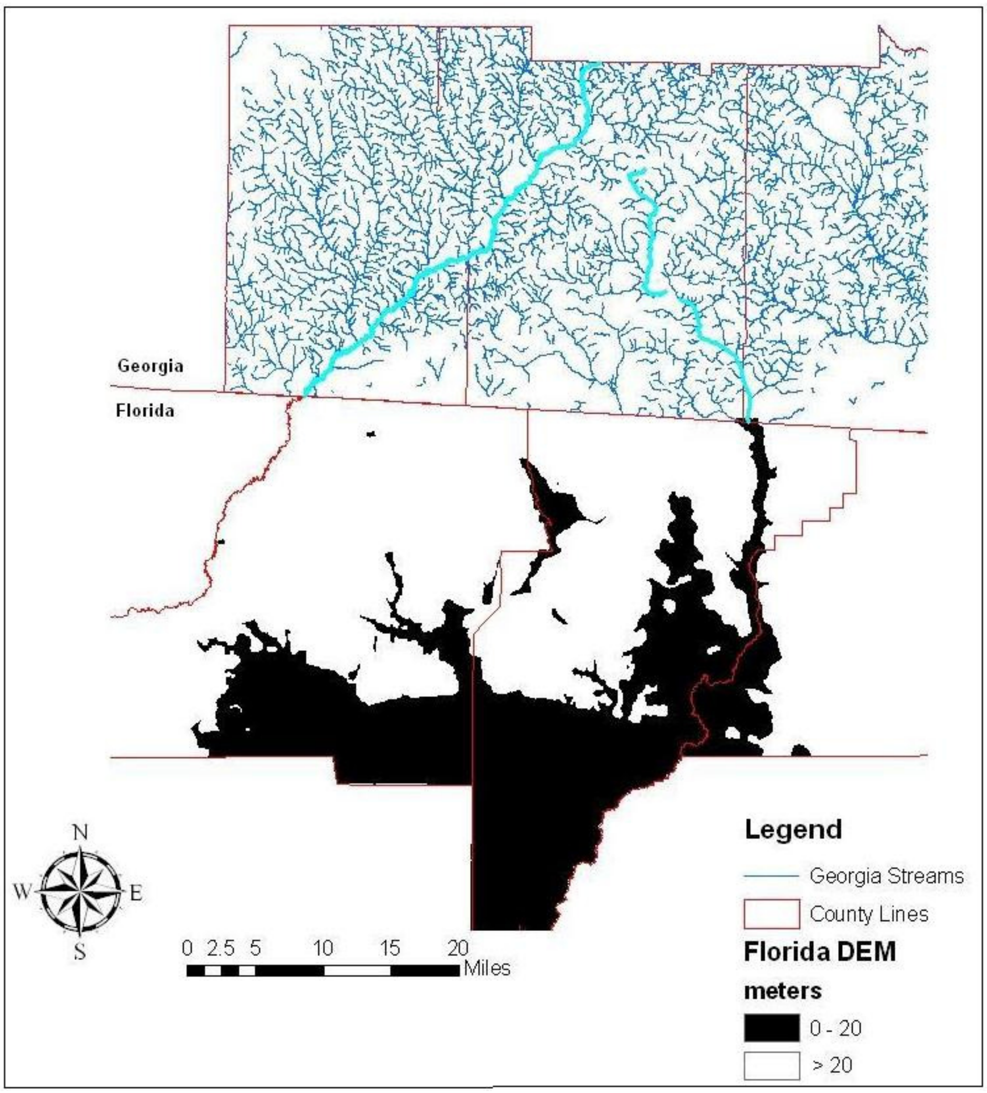

Figure 3.1: Digitizing a boundary for the Red Hills Region.

The resulting boundary created for this research has an area of 280,131 hectares. This is slightly larger than Cox et al.'s description, in which they estimate that the region covers 2,400 km^2 (240,000 ha). Grady County and Thomas County, Georgia contained 6.9 and 25.7 percent of the region, respectively. Jefferson County and Leon Country contained 32.1 and 35.1 percent, respectively. And a minimal 0.1 percent of the region expanded into Brooks County, Georgia (Figure 3.2).

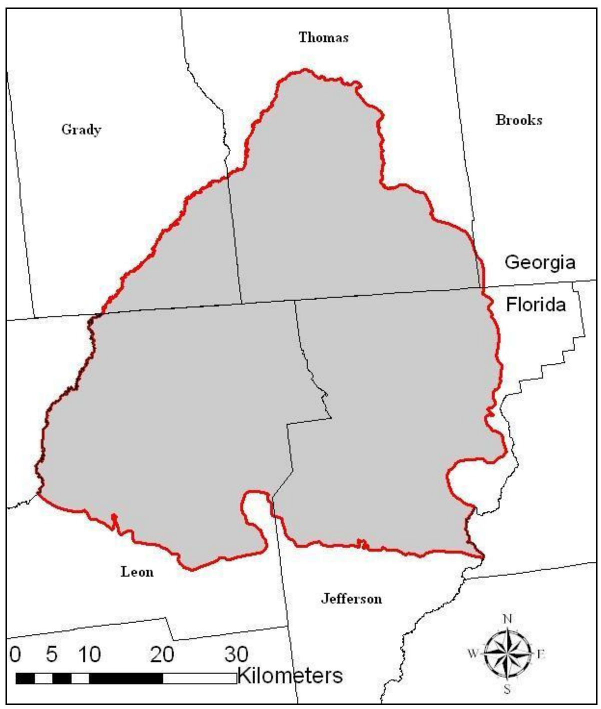

Figure 3.2: The Red Hills boundary

3.2 TYPOLOGY DEVELOPMENT

The development of land cover typology begins with a preliminary survey of available GIS data at the site to determine the basic land cover types present. For this study, the 2001 National Land Cover Data (NLCD) was chosen (NLCD 2001). The NLCD 2001 was produced through a cooperative project conducted by the Multi-Resolution Land Characteristics (MRLC)

Consortium. It is a raster dataset consisting of three main products: 1) per-pixel classified land-cover data (30m pixels); 2) sub-pixel percent imperviousness and; 3) sub-pixel percent tree canopy density (Homer et al. 2004). It was chosen for this study since it was found to be the most recent dataset with a complete and reliable representation of the land cover types present in the Red Hills region. Furthermore, a preliminary review of economic studies (see section 3.3) revealed that ecosystem service coefficients had been documented for cover types present in the NLCD 2001 in a similar context.

The original dataset contained 29 distinct land cover classes. Five of the original land cover classes (Perennial Ice/Snow, Dwarf Scrub, Moss, Paulistrine and Estuarine Wetlands) were excluded entirely from the study, since these were not present in the study area. Additionally, some of the original classes were merged together, since their ecosystem service value estimates would be equal. All three categories of "Developed" land cover (low intensity, medium intensity and high intensity), plus the "barren land" class (negligible presence at 0.05%), were consolidated into a single class and assumed to provide no ecosystem services. The "Shrub/Scrub" class from the original land cover classification also had insignificant presence (0.26%) and was consolidated with the "Grassland/Herbaceous" class. Finally, the "Woody Wetlands" and the "Emergent Herbaceous Wetlands" were consolidated into a single class representing wetland areas. The resulting classification contained ten land cover classes: Open Water; Urban Green Space; Urban Developed; Deciduous Forest; Evergreen Forest; Mixed Forest; Grass/Herbaceous; Pasture; Cropland and; Wetland. A full description of these classes is provided in Appendix A.

Forests dominated the study area, accounting for 53.6% of the 280,000 hectares in the region. Wetlands were the next most predominant class, covering about 16% of the region. These two classes were followed by Grass/Herbaceous, Urban Green Space, Cropland, Pasture, Urban developed, and Open Water. The latter accounted for less than one percent of the total Red Hills region. These proportions are presented in Figure 3.3, which makes clear how natural land covers such as forests, wetlands and grassland are more predominant in the Red Hills when compared to humanly altered land covers such as urban land, cropland and pasture.

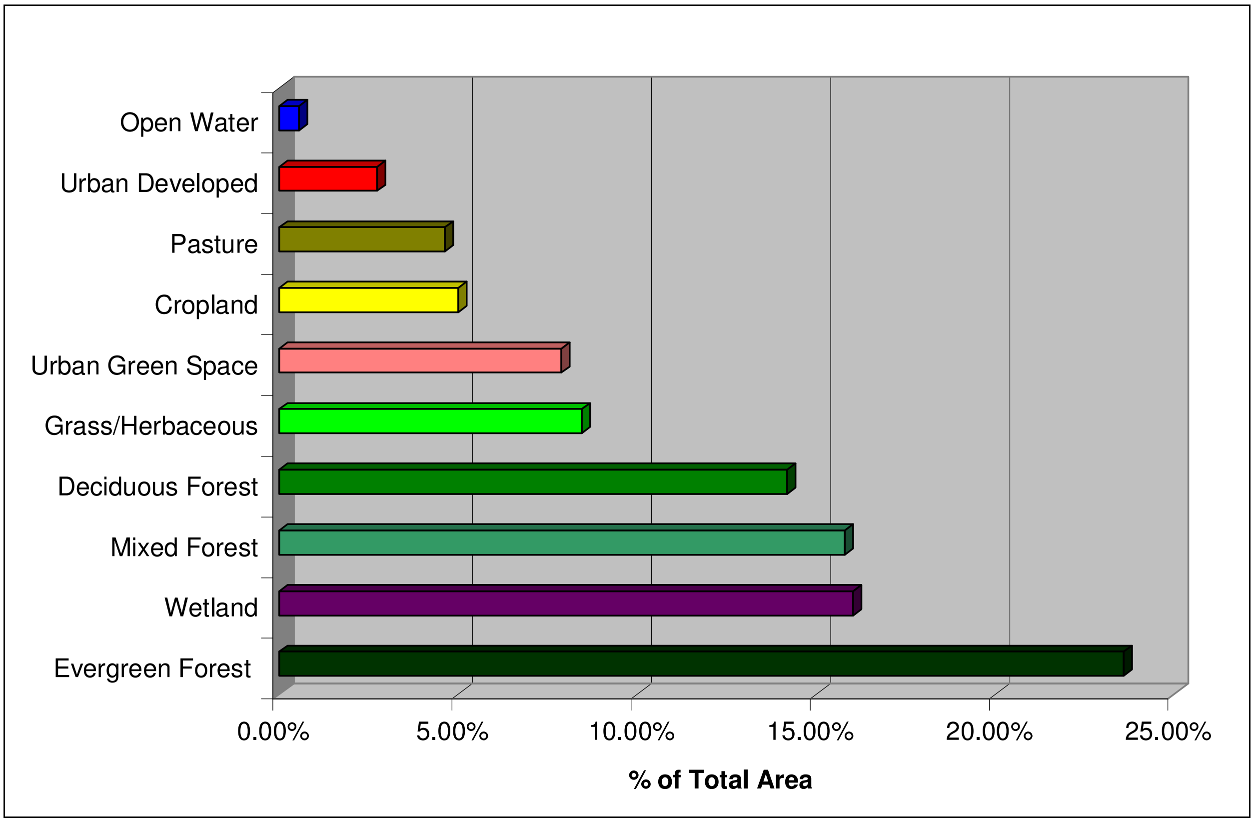

Figure 3.3: Land cover proportions in the Red Hills Region

Other land cover datasets that were considered for use in this study included the National Land Cover Data 1992, and the Coastal Change Analysis Program (C-CAP) Land Cover Data (NOAA 2010). These datasets were originally intended to be used for historical analysis purposes. However, inconsistencies between datasets, as explained in section 3.6, made this historical analysis infeasible.

3.3 LITERATURE SEARCH AND ANALYSIS

During the literature search portion of this study, a number of studies were analyzed in order to select those that had valuation coefficients well suited for the Red Hills Region. Ecosystem service valuation studies that focused on the Southeastern region of the United States were scarce in the literature. There were, however, studies suited for temperate climates, with characteristics similar to the southeastern U.S. so that the valuation coefficients were transferable. Estimates for all cover classes from the proposed typology were therefore successfully produced.

Most relevant to this study were estimates from Liu et al. (2010b), in which they estimated and mapped the value of ecosystem services in the State of New Jersey. This study was found to be both the most comprehensive and most careful in terms of selecting studies from which they derive their valuation coefficients. The authors followed a set decision rules for selecting empirical studies from the published literature that would allow them to estimate with sufficient accuracy the economic value of ecosystem services in their study area. They use three filters intended to avoid generalization errors when performing their analysis. First, they use land cover to ensure the correspondence of ecosystem services between study sites and policy sites. Second, they apply a socio-economic filter, by only selecting studies that refer to a temperate region in North America or Europe to ensure similarity in socio-economic factors (e.g., income, and attitude towards the environment) between these areas and their study site. And third, they apply a uniqueness filter, by only selecting studies that estimate an ecosystem service that is also provided by their study site's natural environment. A full list of the selected studies can be found in Liu et al. (2010).

After reviewing the decision rules followed by the authors in developing their coefficients, they were deemed transferable to Red Hills region. Though regional differences certainly exist between the northeastern and southeastern United States, they are negligible in terms of ecosystem service valuation for the purposes of this study.

Per unit estimates produced by Troy and Wilson (2006) in their study of Massachusetts were also judged to be appropriate for use in this research. Their values are generally higher than those from Liu et al. (2010b), and will serve as upper bound estimates for this study of the Red Hills region. Coefficients derived from Troy and Wilson (2006) will henceforth be referred to as "Coefficients C."

Not included in these either of these studies is a valuable ecosystem service specific to the Red Hills region. The study site for this research provides habitat for the largest population of red-cockaded woodpeckers on private lands and supports an estimated 3-4% of the remaining population of this endangered species. It is widely reported in the literature that red-cockaded woodpeckers nest almost exclusively in softwood trees, and most commonly in longleaf pine (*Pinus palustris*) (Cox et al. 2001; Engstrom and Baker 1995). Forest land cover is therefore divided into three separate categories in this study: deciduous forest, evergreen forest and mixed forest. Coefficients from Liu et al. (2010b) and from Troy and Wilson (2006) are used as baselines for forested land cover. However, a bonus value of $152/ha/yr is added to the evergreen forest class and a half-bonus of $76/ha/yr is added to the mixed forest class, to account for the value of red-cockaded woodpecker habitat provided by these land cover classes. These bonuses were derived from Grado et al. (2009), which estimate that for non-industrial private land owners, the opportunity cost of maintaining mid-high-quality level habitat for red cockaded woodpeckers was $49.30/acre/year (1994 US dollars). This value was converted into 2004 US dollars and used to represent the value of red cockaded woodpecker habitat, an important ecosystem service provided by the Red Hills region.

Table 3.1: Final coefficients by land cover class and coefficient type (2004 US$/ha/yr).

Land Cover Class	Coefficients A	Coefficients B	Coefficients C
Cropland	$59	$2,140	$3,630
Deciduous Forest	$3,170	$3,647	$5,226
Evergreen Forest	$3,322	$3,799	$5,387
Mixed Forest	$3,246	$3,723	$5,307
Grassland/Herbaceous	$30	$190	$209
Pasture	$30	$190	$3,614
Open Water	$1,890	$1,890	$7,810
Urban	$0	$0	$0
Urban Green	$6,111	$6,111	$10,902
Wetland	$21,486	$28,585	$83,121

The final coefficients to be used for all ten land cover classes present in the typology of this study are presented in Table 3.1. Wetlands represent the highest per-unit value of all represented land cover types. This is mostly due to large values associated with water regulation and supply. The next highest per-unit value is associated with the urban green class. Though somewhat surprising, this has been interpreted as a "scarcity effect." Economic theory holds that "green spaces that remain in a fragmented landscape become increasingly valuable to people as places that provide critical recreational opportunities and ameliorate the effects of human development by assimilating the effluent and byproducts of modern urban society." (Wilson and Troy 2003). Forested areas have the next highest ecosystem service value (except with Coefficients C, where open water has higher value than forests), followed by open water, cropland, pasture, grassland/ herbaceous, and finally urban areas which have zero ecosystem service value.

3.4 MAPPING

Map creation involved the combination of input layers from diverse sources to derive the final output maps. Land cover, political boundary, hydrology and satellite imagery layers were all combined in the mapping process. The latter two categories were used mainly as a check for the land cover layer, to ensure that the classifications were accurate enough for the purposes of this study. The political boundaries served to provide spatial reference to the maps, as well as a few more specific boundaries for a few of the scenario analyses (see section 3.6).

3.5 TOTAL VALUE CALCULATION

Once each hectare of the Red Hills region was assigned a cover type, it was then assigned a value multiplier (coefficients from Table 3.1), allowing ecosystem services values to be summed up and cross-tabulated by service and cover type. The total ecosystem services value flow of a given cover type was then calculated by adding up the individual, non-substitutable ecosystem service values associated with that cover type and multiplying by area as given below:

$$V(ES_i) = \sum_{k=1}^{n} A(LU_i) x V(ES_{ki})$$

Where $A(LU_i)$ = area of land cover type (i) and $V(ES_{ki})$ = annual value per hectare for ecosystem service type (k) generated by land cover type (i).

3.6 SCENARIO ANALYSIS

Five basic scenarios were evaluated: 1) 2001 landscape scenario; 2) Undisturbed landscape scenario; 3) Urban growth scenario; 4) Suburban growth scenario and; 5) Agricultural growth scenario.

The original plan of this study included a historical analysis that would evaluate changes in ecosystem services values associated with land cover changes that occurred in the Red Hills region within a ten year time frame. However, data restrictions impeded such an analysis to be carried out completely. While National Land Cover Data from 1992 does exist, and has been one of the most widely used land cover datasets in the United States, direct pixel-to-pixel comparisons between NLCD 1992 and NLCD 2001 is not recommended for several reasons: 1) NLCD 1992 was based on an unsupervised classification algorithm, whereas NLCD 2001 was based on a supervised classification and regression tree algorithm; 2) the spatial resolutions for the DEMs used for terrain corrections were different between NLCD 1992 and NLCD 2001; 3) through impervious surface mapping, NLCD 2001 identified many more roads than could be identified in NLCD 1992, yet most of these roads were present in 1992; 4) NLCD 2001 imagery was corrected for atmospheric effects prior to classification, whereas NLCD 1992 imagery was not, and; 5) land-cover legends differ slightly between the NLCD 1992 and NLCD 2001. These factors result in substantially different pixel-by-pixel labeling in the two dataset, much of which is probably not genuine land-cover change" (EPA 2010). Indeed, when a comparison was attempted between these two datasets, the results produced were highly dubious. According to the data, "open water" accounted for 1.79 percent of total land cover in the Red Hills region in 1992, and dropped to only 0.56 percent in the 2001 dataset. Even more telling were the differences in "urban green space," which went from 0.2 percent in 1992 to 7.89 percent in 2001, "urban developed," which decreased from 7.28 percent to 2.7 percent, and "barren land," which dropped from 11.9 percent in 1992 to an insignificant 0.05 percent in 2001.

When it was clear that the National Land Cover Data would not be suitable for a historic change analysis, another attempt was made using the Coastal Change Analysis Program (C-CAP) Land Cover Data. The C-CAP data included land cover records for three separate years: 1996, 2001 and 2006. However, this dataset was highly inconsistent with NLCD 2001. While the latter indicates that "urban green space" accounts for 7.89 percent of total land cover in the study site, C-CAP data indicated that only 2.99 percent of the region was in that land cover class for the same year. Similarly, NLCD 2001 indicates that wetlands account for 16.03 percent and forests account for 53.6 percent of the Red Hills region, while C-CAP counters with 25.82 percent and 43.87 percent, respectively. Total area and percentage of land cover from each dataset is presented in Appendix C.

When comparing both NLCD 2001 and C-CAP data with satellite imagery from the same time period, NLCD 2001 was deemed to be more accurate, and the C-CAP data was therefore excluded from further analyses. Without the possibility of making reliable comparisons between NLCD 1992 and C-CAP Land Cover Data, the original plan of performing historical analyses was discarded. Instead, an "undisturbed landscape" scenario was created, and more focus was put into growth scenarios, namely agricultural, urban, and sub-urban growth scenarios.

Scenario 1

The "2001 landscape" scenario reflects the actual state of the Red Hills region in 2001, with no changes to the NLCD layer. This scenario served as a baseline from which modifications were made to reflect either growth in urban, suburban or agricultural areas, or in the case of scenario 2, to reflect a supposed original state of the Red Hills region. From there, changes in total ecosystem services values associated with land cover changes could be evaluated for the entire Red Hills region.

Scenario 2

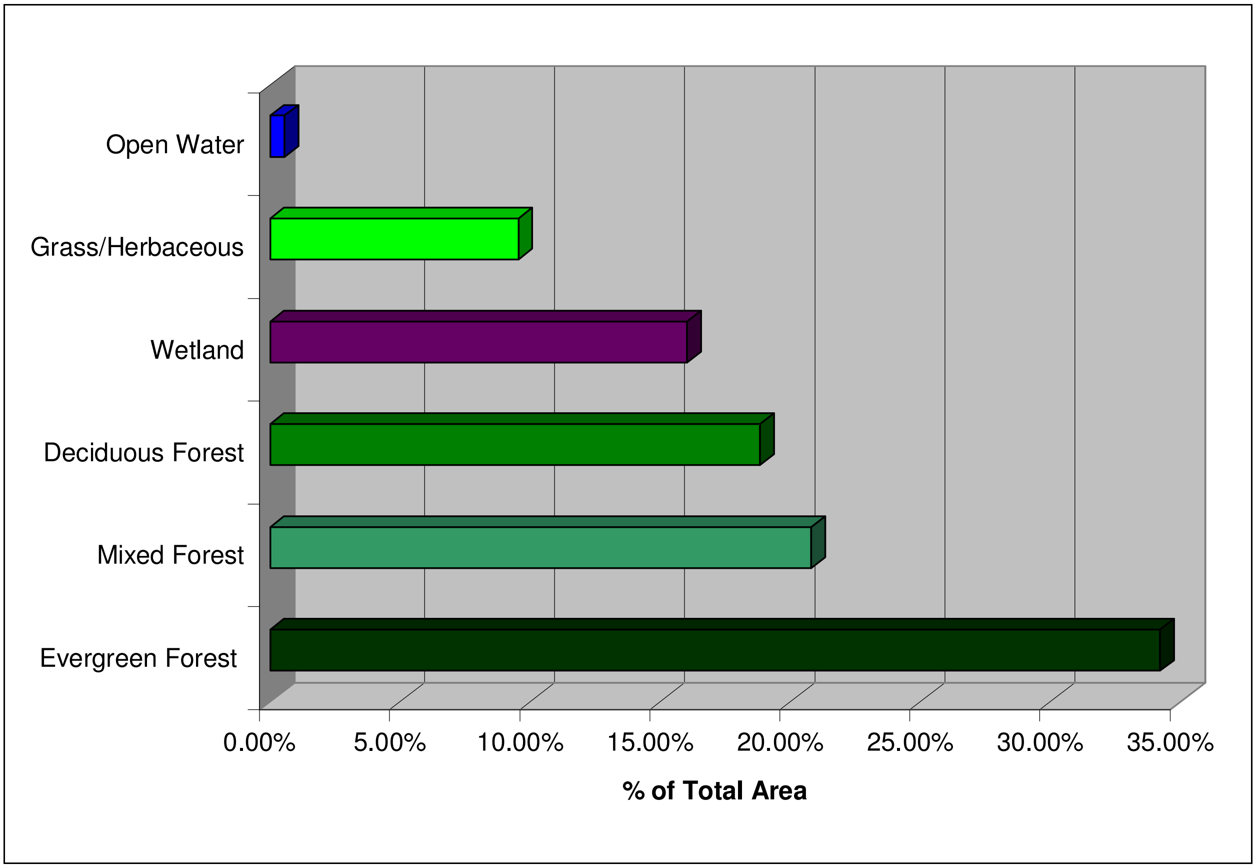

Figure 3.4: Undisturbed land cover proportions

The “undisturbed landscape” scenario reflects what the Red Hills region would look like without any sort of development. It demonstrates the full potential of ecosystem service production in the region’s natural state. For this scenario, cropland, pasture, urban green and urban developed areas from the NLCD 2001 layer were replaced with forestland. Care was taken so that the proportion of deciduous, evergreen and mixed forests was preserved as much as possible when creating this scenario. The NLCD 2001 indicates that 26.49 percent of forests in the Red Hills are deciduous, 44.01 percent are evergreen, and 29.5 percent are mixed. Since the sum of urban green and urban developed classes represented about half of the total land cover that was being changed, those two land cover classes were converted to evergreen forest. Cropland represented about a quarter of the land cover to be changed and was therefore converted to mixed forest. Pasture represented a little less than a quarter of land cover to be altered and was converted to deciduous forest. Figure 3.4 presents percentage of area by land cover class in this scenario.

Scenario 3

The "urban growth" scenario involves an expansion of urban and urban green land cover. For this scenario, neighborhood analysis was used in ArcGIS to simulate the growth of urban areas. Nowak and Walton (2005) project that the proportion of land in the coterminous United States classified as urban will nearly triple from 3.1 percent in 2000 to 8.1 percent in 2050. They estimate that 3.9 percent of land cover will be urban in 2010, 4.8 percent in 2020, 5.8 percent in 2030 and 6.9 percent in 2040. These estimates were used as targets in the neighborhood analysis performed in this step of the research.

The NLCD 2001 was reclassified so that raster cells originally classified as "urban developed" received a value of 100, and all other cells received a value of zero. Neighborhood analyses were performed on the resulting layer, so that a new output raster was created in which each cell value was equal to the mean of its surrounding cells. The result was an output raster in which cells had either a value of zero, or a value higher than zero. Those cells with a value higher than zero were again reclassified into "urban developed." As such, they represented a larger urban area as the original NLCD 2001, and reflected urban growth. Five different neighborhood grid sizes were used in the neighborhood analyses to come up with urban areas that matched the proportions projected by Novak and Walton (2005). Though the exact target percentages were impossible to match due to limitations in the neighborhood analysis, the resulting proportions of urban land cover were close to Nowak and Walton's (2005) projections. Figure 3.4 presents the proportion of each land cover class associated with each growth level, and demonstrates that the "urban developed" and "urban green space" land cover classes both grow at the expense of all other land cover classes in this scenario.

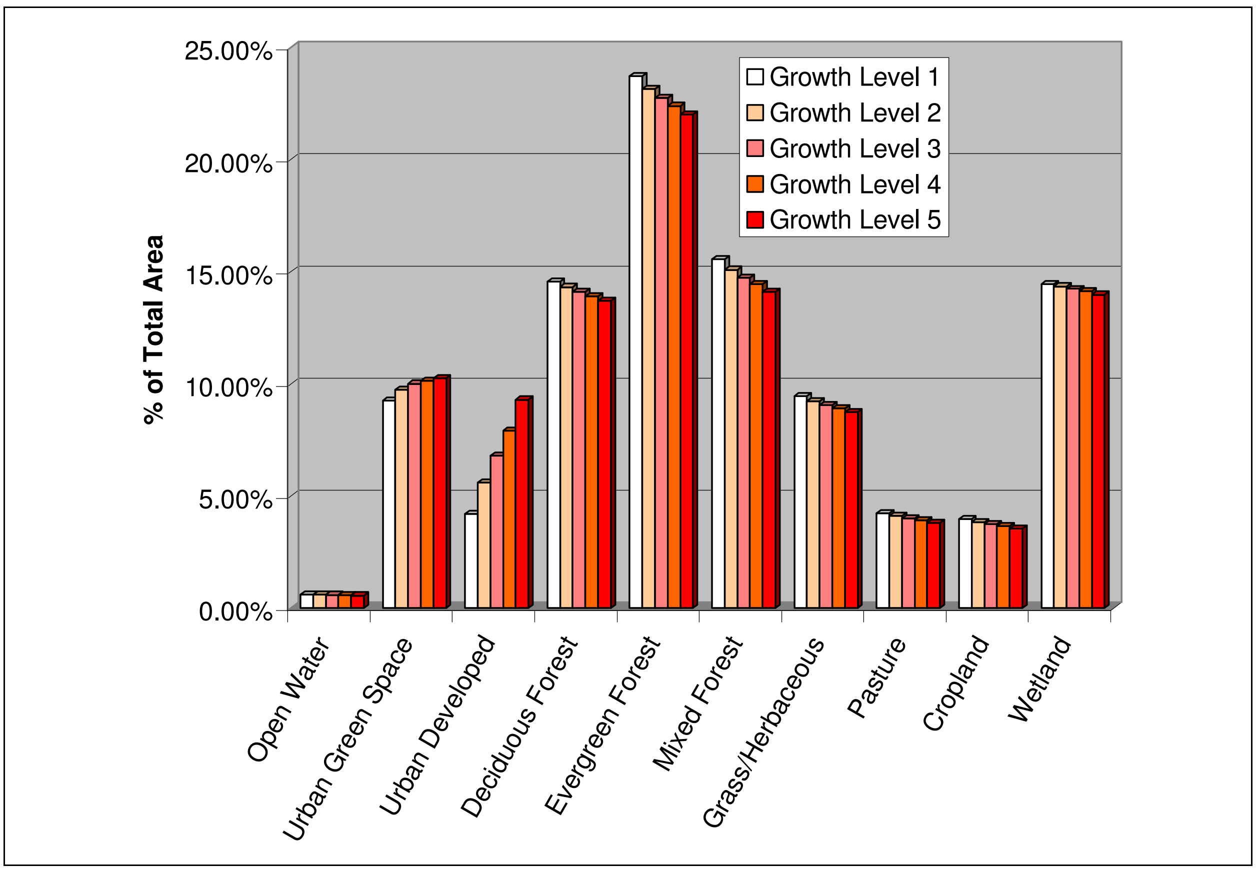

Figure 3.5: Percentage of land cover by urban growth level.

The same neighborhood analysis procedure was performed separately with the "urban green" land cover class. For this land cover class, however, growth was restricted to the urban services area of Tallahassee, Thomasville, and Monticello. Polygons representing the urban services area of these cities were obtained through the property appraiser's office of Leon and Jefferson counties in Florida, and Thomas County in Georgia, and used as a mask for the neighborhood analysis. Parameters in this analysis were adjusted so that the growth of urban green areas were proportional to the growth of urban areas. The urban and urban green growth projections were then combined using Raster Calculator in ArcGIS. When combined, if an "urban" cell overlapped an "urban green" cell, the former was given priority so that urban land cover growth dominated the growth of urban green land cover. This explains why, proportionally, it appears that urban areas grew more than urban green areas in this scenario (Figure 3.5).

Scenario 4

The "suburban growth" scenario was created to illustrate the possible and likely occurrence of urbanization outside of the urban services area of Tallahassee, Thomasville and Monticello. For this scenario, it was assumed that most suburban growth would take place alongside major roads in the Red Hills region, and that it would only occur in areas designated as forested land in the NLCD 2001.

For this analysis, polyline data representing state and interstate highways in the Red Hills region were obtained from state agencies, and then combined in ArcGIS to create a single layer for the entire region. The NLCD 2001 raster layer was then converted into vector data in order to carry out the analysis. Once this was done, land cover polygons from the vector layer that were located within 500 meters of all major highways in the region were selected. This selection was then further constrained to polygons representing forested areas only (deciduous, evergreen and mixed forests). The resulting selection contained 47,363 hectares of forested land that was situated alongside major roads in the region.

The next step was to evaluate the effects suburban growth at three different levels: two, five and ten percent growth. For this, random points were generated within the selected forested area using Hawth's Tools in ArcGIS. Buffers were then placed around the random points, and clipped so that they were restricted by the forested polygon boundaries. Through trial and error, the appropriate radii for the buffers were determined so that the resulting total area of all buffer polygons were equal to two, five and ten percent of the selected forested area. The polygons resulting from these buffers were then combined to the original NLCD 2001 raster layer, and transformed from forested land cover into urban land cover.

If the process of generating random points were to be repeated numerous times, the location of the buffered areas would obviously be different each time. Consequently, the proportion of deciduous, evergreen and mixed forests to be converted into urban land cover would also differ each time. However, within the selected forested area alongside major roads, the total proportion of each forest type is as follows: 24.03 percent deciduous; 49.11 percent evergreen and; 26.86 percent mixed. It can therefore be assumed that if the random point generation process were repeated an infinite number of times, the average proportion of each forest type to be converted to urban land would tend towards the above percentages. Consequently, when analyzing the change in ecosystem services values as a result of suburban growth, these averages were used to represent the proportion of each forest type lost.

Scenario 5

Finally, the “agricultural growth” scenario is intended to reflect and expansion of cropland and pasture into areas that are currently forested. The agricultural growth in this scenario was restricted to an agricultural growth boundary, located to the north of Interstate 10 and to the west of Florida State Highway 59 (Figure 3.6). This area between Tallahassee, Florida and Thomasville, Georgia suffers by far from the most development pressure in the Red Hills region, and is therefore an area of great concern to land owners and policy makers.

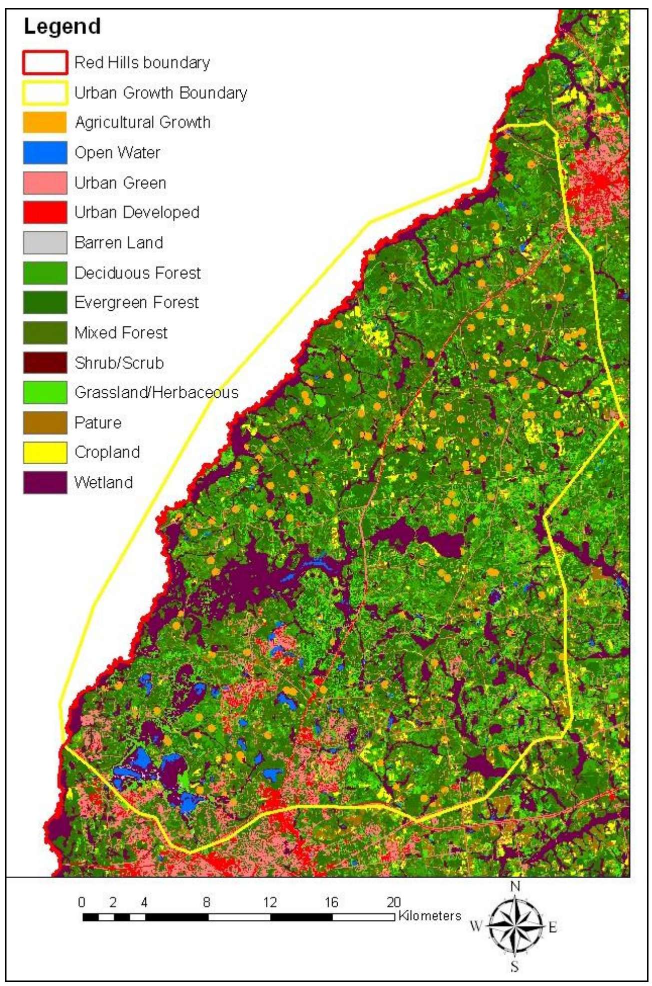

Figure 3.6: The agricultural growth boundary

For this scenario, land cover polygons from the NLCD 2001 vector layer representing forested areas located completely within the agricultural growth boundary were selected. The same procedure from the suburban growth scenario was then employed. Random points were generated, and buffers were created around them and clipped with the forested polygons. The radii again were chosen by trial and error, such that the total area of all buffers represented two, five and ten percent of the total area in the agricultural growth boundary. Within the boundary, the proportion of each forest type is as follows: 26.64 percent deciduous, 46.46 percent evergreen and 26.9 percent mixed. Consequently, when analyzing the change in ecosystem services values as a result of agricultural growth, these percentages were used to represent the proportion of each forest type lost. However, unlike the suburban growth scenario, in which forests were replaced with urban land cover, forestland here was replaced evenly with cropland and pasture.

CHAPTER 4

RESULTS

This chapter will present the results derived from following the decision framework established in the methodology section of this study. These results include: 1) The total value of ecosystem services produced by the Red Hills region; 2) The total value of ecosystem services provided by each land cover class in the Red Hills region and; 3) Changes in total ecosystem service values produced by the Red Hills region, resulting from changes in land cover across the study site.

4.1 TOTAL VALUE OF ECOSYSTEM SERVICES

The land cover distribution of the Red Hills region from NLCD 2001 is shown in Figure 4.1. This figure represents Scenario 1 of our analysis, which serves as a baseline for the subsequent scenarios. While the total area and proportion of each land cover class have already been presented in Figure 3.3, visualizing this distribution in a spatially explicit manner allows us to gain a sense of where each land cover class falls within the region and, consequently, which parts of the region have the most value in terms of flow of ecosystem services.

The red pixels in the southeast portion of the map represent the city of Tallahassee, Florida, and those in the northernmost portion represent Thomasville, Georgia. The corridor between these two urban areas suffers the most development pressure in the Red Hills region. However, there is still clearly a great deal of forested land, as well as valuable wetland ecosystems in that area. Understanding how land cover change affects the value of that land is therefore important, and of interest to stakeholders in the region.

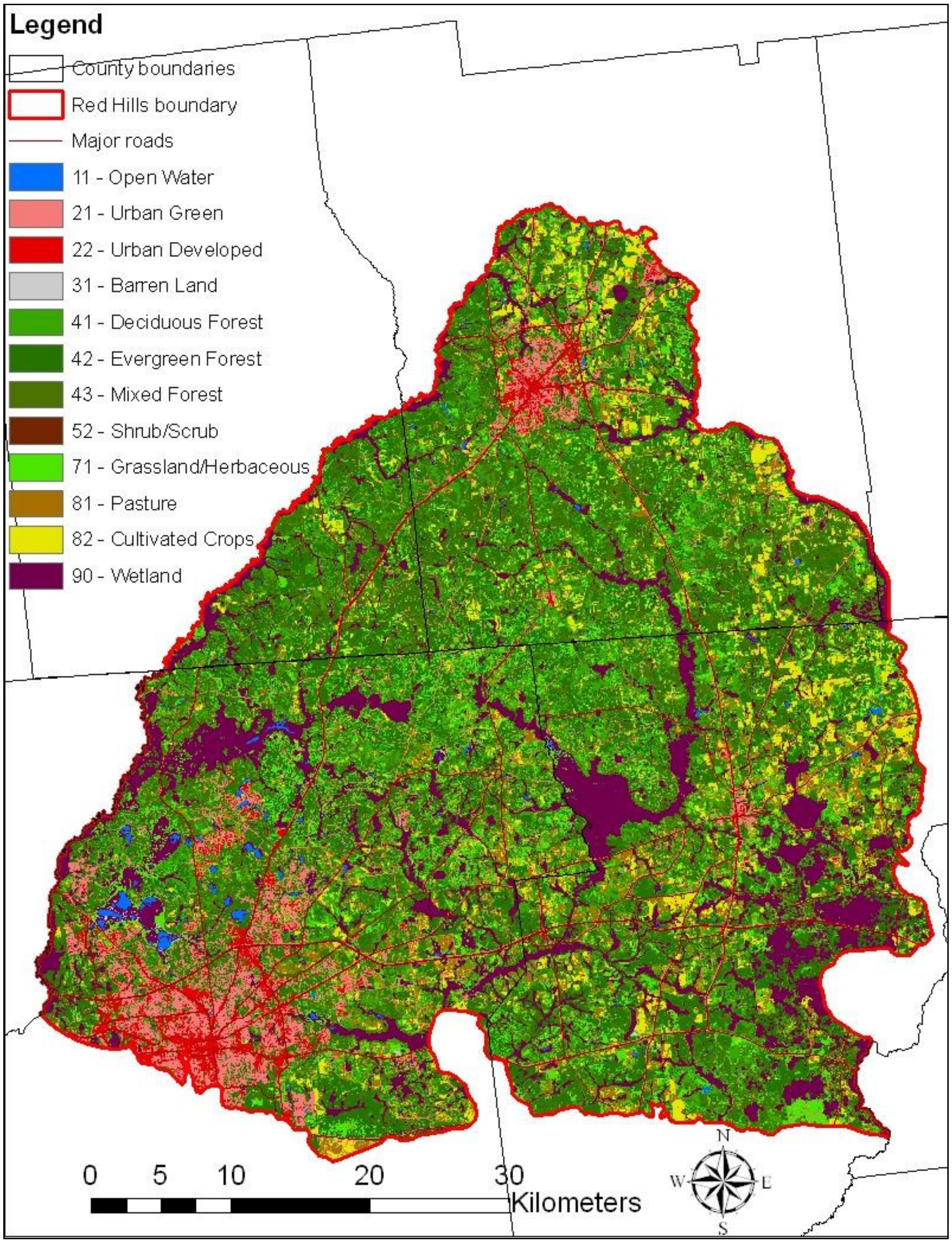

Figure 4.1: 2001 Red Hills Region Land Cover Distribution

This study found the total value of ecosystem services produced by the Red Hills region to be within the range of $1.59 and $4.89 billion (2004 US$) per year. The lower estimate is a result of benefit transfer analysis using coefficients A. With coefficients B, the total value increases to $2.02 billion. And the higher estimate of $4.89 billion was a result of benefit transfer using coefficients C.

Table 4.1: Total value of ecosystem services provided by the Red Hills region (2004 US$/yr)

Land Cover Classes	*Area (ha)*	*Total ESV flow (Coefficients A)*	*Total ESV flow (Coefficients B)*	*Total ESV flow (Coefficients C)*
Cropland	14,000	$826,000	$29,959,104	$50,820,000
Deciduous Forest	39,784	$126,115,280	$145,103,436	$207,911,184
Evergreen Forest	66,078	$219,511,116	$251,030,322	$355,962,186
Mixed Forest	44,287	$143,755,602	$164,880,501	$235,031,109
Grassland/ Herbaceous	26,746	$802,380	$5,089,000	$5,589,914
Pasture	12,987	$389,610	$2,471,055	$46,935,018
Open Water	1,565	$2,957,850	$2,958,412	$12,222,650
Urban	7,685	$0	$0	$0
Urban Green	22,103	$135,071,433	$135,069,793	$240,966,906
Wetland	44,894	$964,592,484	$1,283,303,784	$3,731,634,174
TOTAL	280,131	$1,594,021,755	$2,019,865,407	$4,887,073,141

Table 4.1 separates ecosystem service value flows by land cover class, and summarizes the total values from each analysis. The results show that the largest cumulative economic values are produced by ecosystem services associated with wetlands and forested areas, which is consistent with the per-unit data reported in tables 2.6 and 2.8.

4.2 CHANGES IN ECOSYSTEM SERVICE VALUES

The final portion of this study consisted of analyzing how changes in the 2001 land cover would affect the total value of ecosystem services provided by the Red Hills region. The first changes that we analyzed were from Scenario 2, the “undisturbed scenario”, which is illustrated in Figure 4.2.

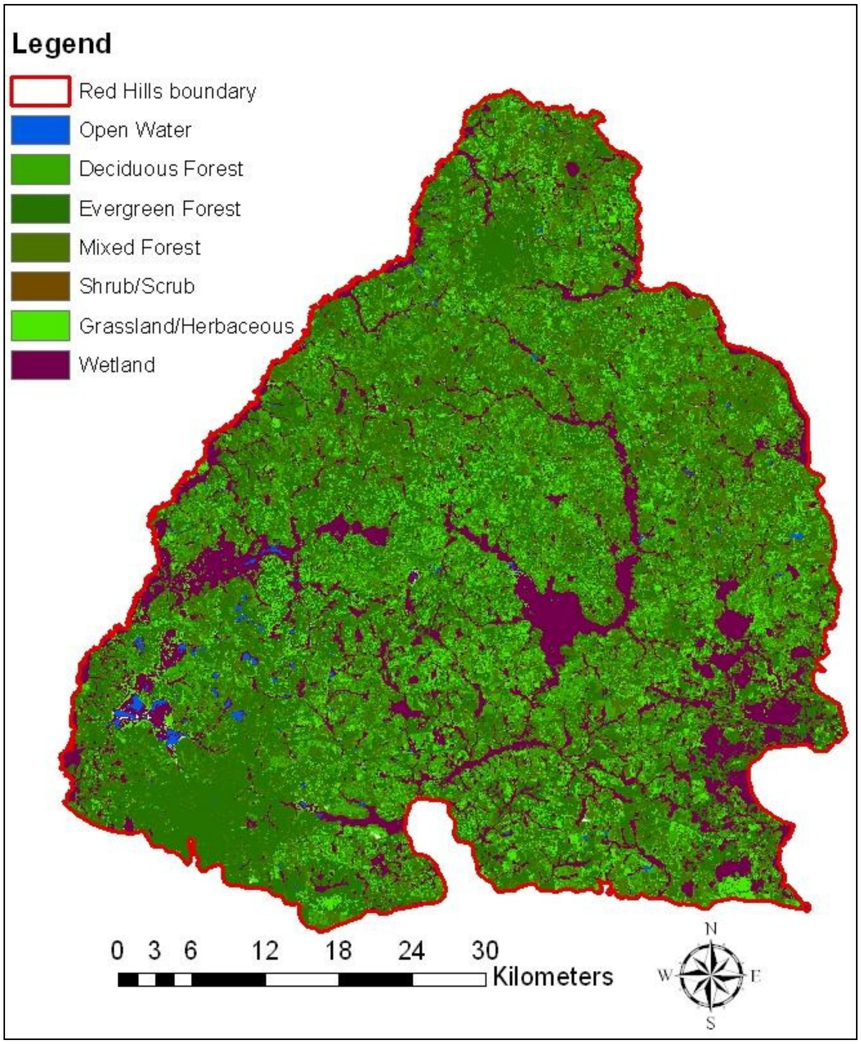

Figure 4.2: Undisturbed Scenario Land Cover Distribution

Table 4.2 provides ecosystem service value flows from the “undisturbed scenario.” Using coefficients A, results indicate that a Red Hills region devoid of human activity produces ecosystem services worth $49.3 million more than those produced in its 2001 state. Coefficients B produced a similar increase of $45.2 million. These values are somewhat lower than what one might expect, since human activity is generally thought to decrease ecosystem services values. Even more surprisingly, coefficients C lead to an undisturbed scenario that produces a $36.1 million loss in value of ecosystem services. The reason for this these results becomes apparent when one considers that for all three sets of estimates, the value of urban green land cover is substantially higher than that of forestland. Consequently, changing urban green land cover into forestland results in a net loss in ecosystem service value. Moreover, Troy and Wilson's (2006) estimates of ecosystem services provided by cropland and pasture are much closer to those provided by forestland, when compared to Liu et al.'s (2010) estimates. Therefore, with coefficients C, the gains obtained when switching from pasture and cropland to forestland are not enough to offset the losses that result from switching urban green to forestland.

Table 4.2: Total value of ecosystem services for the undisturbed scenario (2004 US$/yr)

Land Cover Classes	*Area (ha)*	*Total ESV flow (Coefficients A)*	*Total ESV flow (Coefficients B)*	*Total ESV flow (Coefficients C)*
Open Water	1,565	$2,957,850	$2,957,850	$12,222,650
Deciduous Forest	52,772	$167,287,240	$192,474,324	$275,786,472
Evergreen Forest	95,866	$318,466,852	$364,194,934	$516,430,142
Mixed Forest	58,287	$189,199,602	$217,002,501	$309,329,109
Grass/Herbaceous	26,746	$802,380	$5,089,000	$5,589,914
Wetland	44,894	$964,592,484	$1,283,303,784	$3,731,634,174
TOTAL	280,130	$1,643,306,408	$2,065,022,394	$4,850,992,461

The third scenario analyzed five different levels of urban growth within the Red Hills region. Figure 4.3 illustrates the results obtained from the growth model. In 2001, urban developed land represented 2.74 percent of all land cover in the Red Hills region. Growth level 1 represents the Red Hills with 4.2 percent urban developed land; at level 2, 5.59 percent is urban developed; 6.8 percent is urban developed at level 3; 7.91 percent at level 4 and; 9.3 percent at level 5. In all five levels, there is also proportional growth in “urban green” land cover. However, when urban growth overlapped with urban green growth, the former was given priority, so that it trumped the growth of urban green.

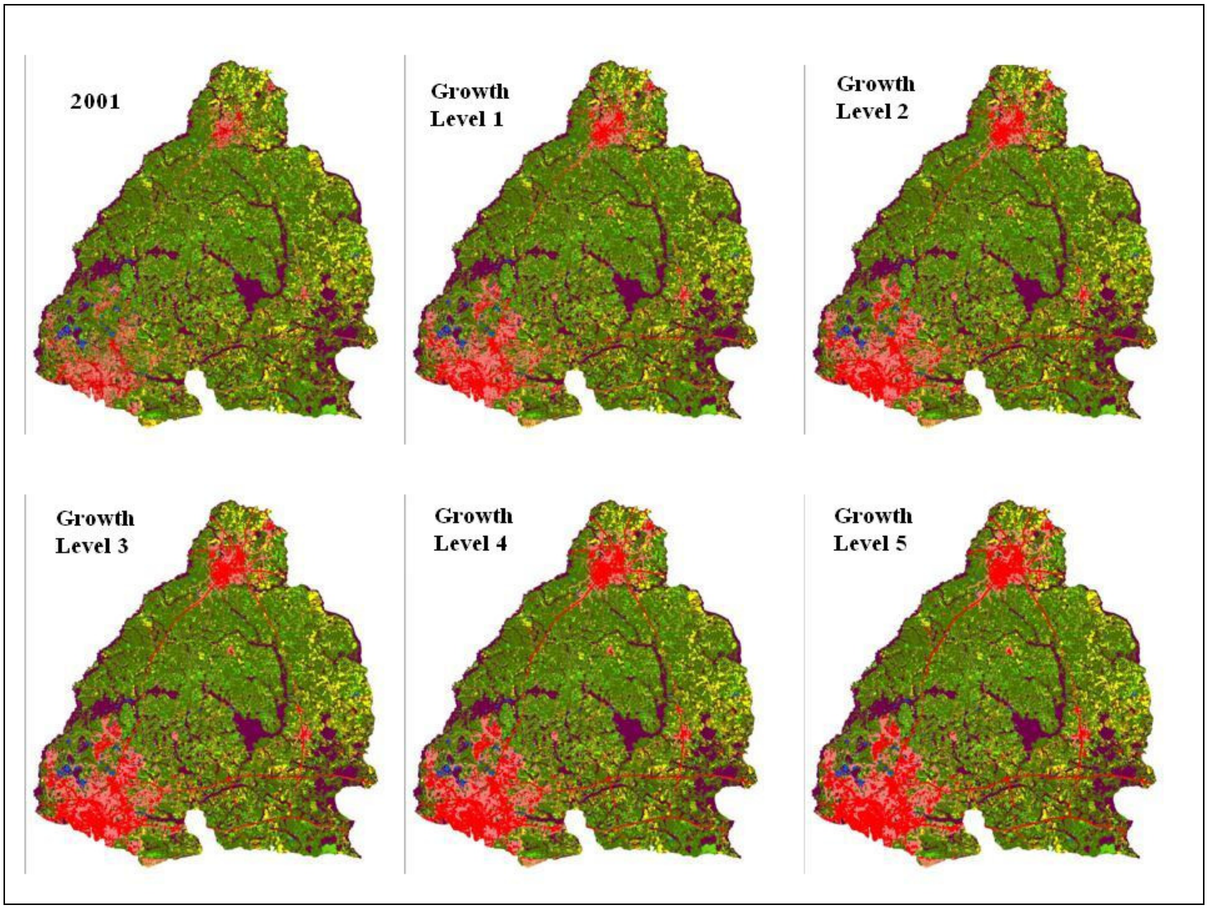

Figure 4.3: Urban Growth Scenario

Changes in the total value of ecosystem services for the Red Hills region at each growth level are presented in Table 4.3. All values from the table represent a loss of ecosystem service value. As expected, higher levels of urban area are associated with higher losses. This holds true regardless of the type of coefficient that is used. With coefficients A, the loss in value ranges from \$311 to \$352 million per year from growth level 1 to growth level 5. Using coefficients B, the losses range from $410 to $466 million. Coefficients C result in the highest losses, ranging from $1.06 to $1.18 billion. This trend is also expected since urban developed land has a null value in terms of ecosystem services provided, and coefficients C are larger than coefficients B, which are larger than A.

Table 4.3: Losses in ecosystem service value associated with urban growth levels and coefficient type (2004 US$/yr)

	Growth Level 1	Growth Level 2	Growth Level 3	Growth Level 4	Growth Level 5
Coefficients A	$310,963,659	$319,198,192	$328,604,066	$338,503,179	$351,896,328
Coefficients B	$409,778,110	$422,047,533	$434,957,370	$448,095,865	$465,594,308
Coefficients C	$1,056,691,879	$1,083,592,940	$1,112,492,003	$1,142,334,344	$1,182,541,097

In scenario 4, suburban growth alongside major roads in the Red Hills region was analyzed. All forested land located within 500 meters of a major road were selected, and converted to urban land cover at three different levels: two, five and ten percent. Figure 4.4 illustrates the change at a 5 percent level. The map on the left shows the areas that were converted, and the map on the right illustrates the converted land after being incorporated with the remaining land cover classes.

Since urban land cover has a null value in terms of ecosystem service flow, the conversion of forested land into urban land will result in loss equivalent to the value of ecosystem services being provided by those forests. Looking back at Table 3.1, it can be concluded that when using Coefficients A, every hectare of deciduous forest converted into urban land is associated with a $3,170 loss in ecosystem service value. Similarly, when using Coefficients B, every hectare of evergreen forest converted into urban is associated with $3,799 in losses. And With Coefficients C, mixed forestland that is converted into urban land leads to losses worth $5,307.

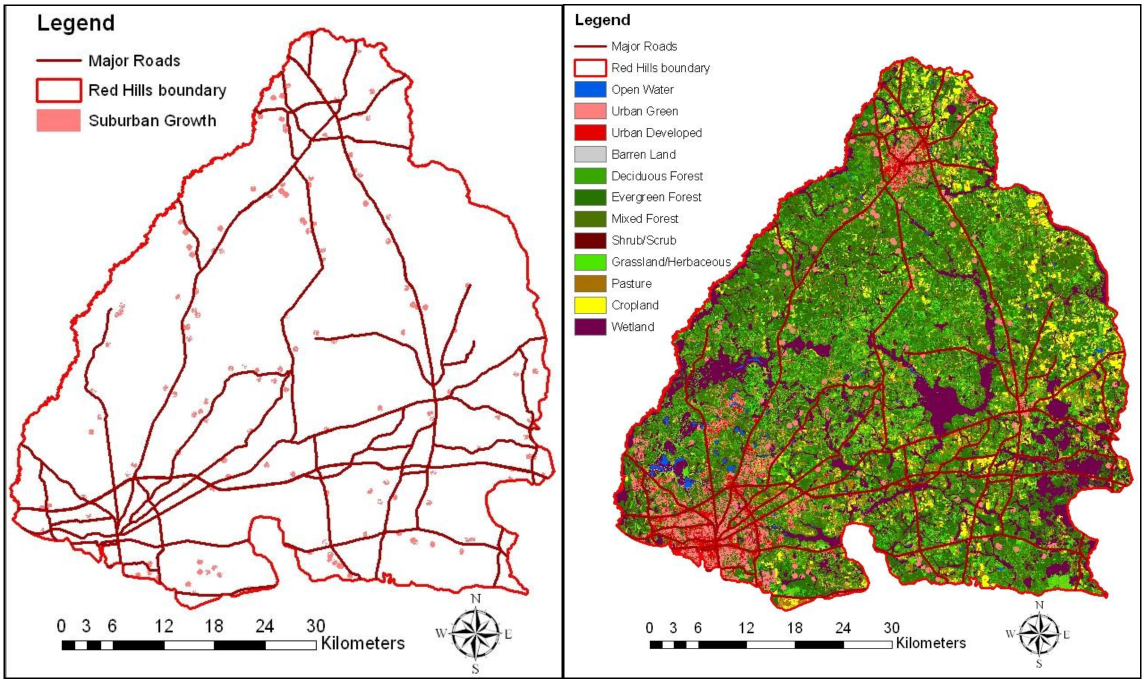

Figure 4.4: Suburban Growth Scenario

Table 4.4 presents the total losses associated with each suburban conversion level and with each coefficient type. A two percent conversion rate results in a total loss of $3.1 million when using Coefficients A. The losses are proportional at $7.7 million with a five percent conversion rate, and at $15.4 million with a ten percent rate. Using Coefficients B and C, the losses are obviously greater, since they both attribute more value to forested land when compared to Coefficients A.

Table 4.4: Losses in ecosystem service values associated with suburban growth levels and coefficient type. (2004 US$/yr)

Conversion Rate	Forest Type	ha	Coefficients A	Coefficients B	Coefficients C
2%	Deciduous	228	$722,760	$831,580	$1,191,528
	Evergreen	465	$1,544,730	$1,766,535	$2,504,955
	Mixed	254	$824,484	$945,642	$1,347,978
		Total	$3,091,974	$3,543,757	$5,044,461
5%	Deciduous	569	$1,803,730	$2,075,303	$2,973,594
	Evergreen	1163	$3,863,486	$4,418,237	$6,265,081
	Mixed	636	$2,064,456	$2,367,828	$3,375,252
		Total	$7,731,672	$8,861,368	$12,613,927
10%	Deciduous	1138	$3,607,460	$4,150,606	$5,947,188
	Evergreen	2326	$7,726,972	$8,836,474	$12,530,162
	Mixed	1272	$4,128,912	$4,735,656	$6,750,504
		Total	$15,463,344	$17,722,736	$25,227,854

The fifth and final scenario that was analyzed involved the growth of agricultural land cover. In this scenario, conversion of forest land into crops and pasture was restricted to a portion of the Red Hills region that is particularly susceptible to development and conversion pressure. This area was illustrated in Figure 3.6 and defined as the agricultural growth boundary. In this analysis, forested land within the agricultural growth boundary was converted at three different rates: two, five and ten percent. Half of the land in each analysis was converted into pasture, while the other half was converted into cropland. Figure 4.5 illustrates the conversion of five percent of forested land within the boundary into pasture and cropland.

In the case of agricultural growth, the conversion of forestland does not result in a total loss of ecosystem service value, since there are also values associated with cropland and pasture. Examining Table 3.1, losses associated with the conversion of each hectare of forestland can be obtained. For example, when using Coefficients A, every hectare of deciduous forest converted into cropland results in a net loss of $3,111, while the conversion into pasture results in a net loss of $3,140. Coefficients B indicate that every hectare of evergreen forests lost to cropland is associated with $1,659 in losses, and every hectare lost to pasture is associated with $4,457 in losses. And with Coefficients C, each hectare of mixed forest lost to cropland leads to a loss of $1,677 in ecosystem service value, while each hectare lost to pasture leads to $1,693 in losses.

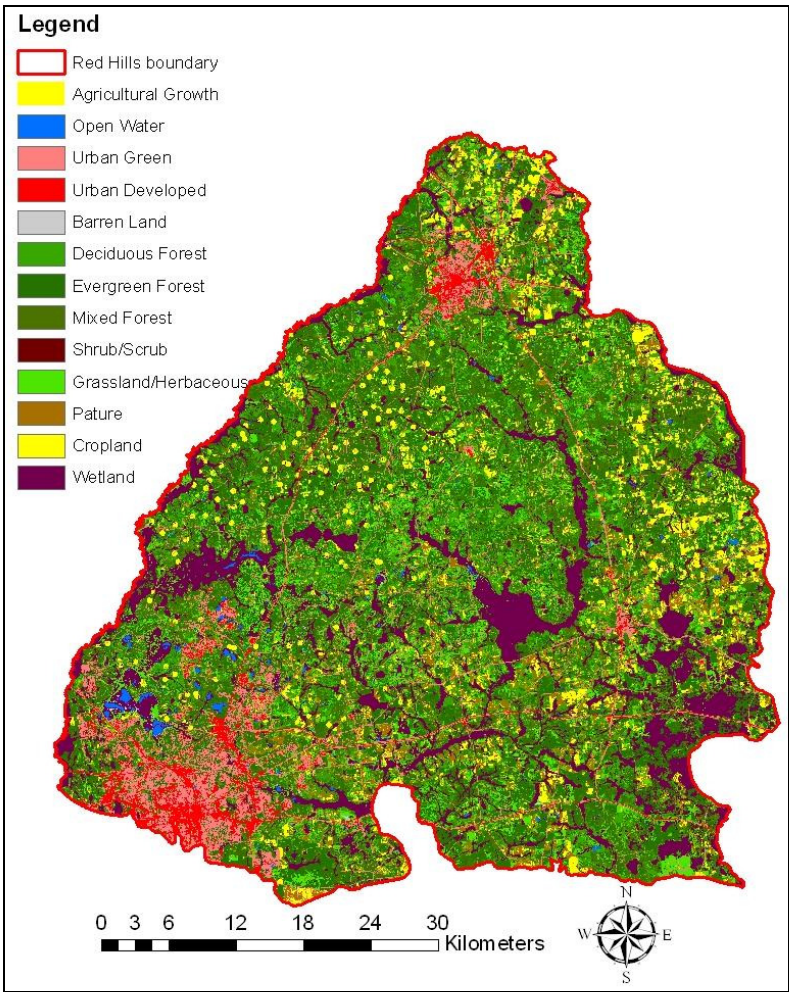

Figure 4.5: Agricultural Growth Scenario

Table 4.5: Losses in ecosystem service values associated with suburban growth levels and coefficient type. (2004 US$/yr)

Conversion Rate	ha	Coefficients A	Coefficients B	Coefficients C
2%	1,103	$3,547,823	$2,838,010	$1,875,701
5%	2,758	$8,871,233	$7,096,381	$4,690,175
10%	5,517	$17,745,744	$14,195,395	$9,382,115

Table 4.5 presents the total losses associated with each agricultural conversion level and with each coefficient type. At a two percent conversion rate, total losses originating from the conversion of forestland into cropland within the urban growth boundary ranges from $1.87 million when using Coefficients C, to $3.55 million when using coefficients A. A ten percent conversion rate leads to losses ranging from $9.38 million to $17.75 million. In this scenario, the use of Coefficients C result in smaller losses associated with agricultural growth, since the estimates from Troy and Wilson (2006) place a higher value on ecosystem services provided by cropland and pasture, when compared to estimates from Liu et al. (2010b).

CHAPTER 5

DISCUSSION

5.1 SUMMARY AND DISCUSSION OF RESULTS

This study estimates that the value of ecosystem services provided by the Red Hills region falls within the range of $1.6 and $4.9 billion per year, which is consistent with estimates from the literature that valued similar areas. The state of New Jersey, which was the study site of Liu et al. (2010), has an area eight times larger than that of the Red Hills, and the authors estimate that ecosystem services provided by the entire state ranges between $11.6 and $19.6 billion (2004 US$) per year. Troy and Wilson (2006) looked at ecosystem services provided by Massachusetts, which is roughly ten times larger than the Red Hills region, and estimated that they provide approximately $7.4 billion (2004 US$) per year in value. Finally, Williams et al. (2003) estimated the total value of ecosystem services provided by Scotland, which is 28 times the size of the Red Hills, to be equal to $25 billion (2004 US$) per year.

As expected, most of the ecosystem service value provided by the Red Hills region comes from wetland and forested areas. Not only do these two land cover classes produce the highest per hectare value, they also cover most of the landscape in the region. Forests and wetlands combined account for 69.6 percent of the Red Hills landscape, and approximately 92 percent of total ecosystem service value provided by the region. Thus, it is important to take the values provided by this study into account, particularly when making land use decisions that would affect forested and wetland areas.

The results from the undisturbed scenario indicate that converting humanly altered land cover (i.e. urban developed, urban green, pasture and cropland) into forestland produces $49.3 million in gains using Coefficients A, and $36.1 million in losses when using Coefficients C. These values are surprising, since we expected that converting the 2001 Red Hills landscape into an undisturbed landscape would produce a significantly higher total value for the region. And examination of Table 3.1 helps explain the low values. Only 20 percent of land cover was actually converted in this scenario, and over one third of it fell within the "urban green" class. This land cover class has a per hectare value coefficient that is almost twice as large as the average forest land cover coefficient, both for Coefficients A and C. This is due to the high recreational value associated with urban green land cover, which is generally located around developed areas where green land tends to be scarce. This scarcity leads people to place higher premiums on this type of land, both for recreational and aesthetic purposes. As a result of this, converting urban green land into forestland in the undisturbed scenario resulted in a reduction of roughly 50 percent in terms of ecosystem service value. Using Coefficients A, this loss was offset by the gains resulting from the conversion of agricultural lands (pasture and cropland) into forests, and therefore a net gain of $49.3 million was realized. However, with Coefficients C, the per hectare value of agricultural lands are much closer to those of forestland. Consequently, converting pasture and cropland into forests did not result in gains large enough to offset the losses from converting urban green into forests, and the result was a net loss of $36.1 million.

The third scenario evaluates the effect that urban growth has on the total value of ecosystem services provided by the Red Hills region. Five levels of growth, matching decadal urban growth projections for the United States produced by Nowak and Walton (2005), were analyzed. It should be noted that these levels represent the projected growth for the entire coterminous United States. The authors point out that most of the urban growth is projected to occur around the more heavily urbanized areas of the country, which is not the case of the Red Hills. An analysis of C-CAP land cover data (NOAA 2010) indicates that between 1996 and 2006, urban developed land cover in the study site increased by just 0.2 percentage points. The growth levels in this scenario are therefore not intended to project the growth in the region between 2001 and 2050. They represent possible urban growth without a specific timeline, and the changes in ecosystem service values associated with this growth. Considering the growth between 1996 and 2006, it is likely that within the next fifty years the Red Hills region will only reach “Growth Level 1” from this scenario. But even this growth level results in substantial losses in ecosystem service values, within the range of $311 million and $1.05 billion per year.

The fourth and fifth scenarios involve the conversion of forestland into either urban or agricultural land. In both scenarios, the result of land cover change was always a loss in total ecosystem service value, since urban developed land, cropland and pasture all have lower per hectare coefficients than forestland. The analyses done in these two scenarios highlight the usefulness of the coefficients provided in Table 3.1, which can be used to assess the change in values associated with converting one hectare of one land cover class, such as forests, into another land cover class, such as pasture.

5.2 STUDY IMPLICATIONS

Although they often go unnoticed, ecosystems provide numerous services that are extremely valuable to humans. As we continue to develop and change the land around us, it is important that we add the value provided by ecosystem services into the decision making equation. This study is a step forward in this process. It provides an estimate of the total value ecosystem services provided by a region known for its tremendous natural assets, yet that suffers strong development pressure from surrounding urban areas. It also demonstrates the losses that may occur when converting land cover that has high ecosystem service values, such as forests and wetlands, into land cover that produces less ecosystem services value, such as urban or agricultural land.

David Pearce, in his forward to "Applied Environmental Economics" (Bateman et al. 2003), states that "viewed from a global perspective, there is a one-to-one relationship between the decline of forested land and the increase in land devoted to crops and pasture. The factors giving rise to land use change are many and varied. But one of the most powerful is the comparative economic returns to 'converted' land relative to the economic returns to 'natural' land. In short, the issue is conservation versus conversion, and this is a conflict that is invariably resolved in the favour of conversion." By making the natural value of land more explicit, it is much more likely that decisions will lean more towards the preservation of natural land and the services that they provide, as opposed to their conversion into something that is seen as more economically profitable. The estimates from this study can be used to assess the most valuable areas of the Red Hills region. Troy and Wilson (2006) and Liu et al. (2010) divide their study sites into tributary basins, and then use their estimates to gain a sense of the average value of each watershed in the region. Alternatively, the estimates can be used at a parcel level, and factored into a cost-benefit analysis for development projects. This way, a looking to build on a specific set of parcels can better assess the gains and losses resulting from his development proposal.

While previous research has shown that there are some problems with the practice of environmental valuation, this should not deter researchers from improving the valuation literature by producing more studies. The continued efforts by economists, ecologists and other over the past several years to provide economic value to ecosystem services have come a long way in helping to place the natural environment at the forefront of land use decisions. And the continued growth of this field of study will only result in more accurate representations of the true value of ecosystems.

5.3 LIMITATIONS AND FUTURE STUDIES

As with most studies that involve benefit transfer analysis, this research has limitations that need to be acknowledged. The accuracy of estimates used to value ecosystem services in the Red Hills region is one aspect that needs to be addressed. There are three factors that may affect the accuracy of benefit transfer estimates: 1) the accuracy of the primary studies when performing their analyses and deriving value estimates; 2) the comprehensiveness of ecosystem services included in the research and associated with each land cover class and; 3) the similarity between study sites in the original studies and the final study.

Regarding the first factor, primary studies rarely produce per unit values (e.g. $/ha/yr) of ecosystem services. Most values produced by these studies are much more specific, and therefore much more difficult to use in benefit transfer. A couple of examples are studies that come up with a value that people would pay to protect a certain species, or studies that determine how much people would to compensate the loss of natural area. In order to transform these values into something that can be used in benefit transfer, researchers must sometimes make decisions which may be contested by others. Often researchers are faced with the decision of generalizing a certain estimate, or having no valuation estimate at all. For instance, some values may be development dependent, that is, there may be greater recreational value in areas closer to development than in areas farther away. But it may be difficult to adequately quantify these changes in value for a specific value, and so the researcher may choose to utilize an average value for the entire area. It is therefore important to document the decisions made when coming up with valuation estimates, so that they can be analyzed by researchers who intend to use them for benefit transfer.

The second factor refers to limited availability of studies that value certain ecosystem services. Though the field of valuation is rapidly growing, there are still many landscapes of interest from an environmental perspective that simply have not yet been studied adequately for their non-market ecosystem service values (Liu et al. 2010b). Such is the case for many of the landscapes in the Red Hills. For example, Coefficients A do not include a value for soil retention and formation in forestland, because there were no studies that matched the selection criteria in order to be used in benefit transfer. Nutrient regulation in wetlands is another example of a services provided by a land cover that is not accounted for. When performing benefit transfer, it is therefore important to point out that the availability of valuation studies limits the accuracy of estimates. In fact, this gap in valuation points leads most valuation studies to describe their results as “conservative estimates” (Costanza et al. 1997b, Troy and Wilson 2006).

The third factor concerns the transferability of estimates from one study site to another. While efforts were made to transfer estimates from regions similar to this study site, they are not perfectly tailored for the Red Hills region. However, care was taken to utilize the most recent and comprehensive valuation studies in the benefit transfer portion of this study. And the red cockaded woodpecker habitat was incorporated into the estimates in an effort to account for this unique service provided by the red hills region.

Another shortcoming of this study was a result of the lack of consistent data suitable for a historic land use change analysis. While it is clear that GIS have come a long way in recent years, and that land cover datasets are more and more detailed, it is still difficult to obtain reliable data at a scale that is relevant for regional analysis, and which allows for both retrospective analysis (historic change) and future projections. With the increased availability of GIS data, and with a more thorough analysis of past trends in terms of land cover changes, future studies improve on the scenario analyses conducted in this study. The urban growth scenario, for instance, was conducted using neighborhood analysis in ArcGIS and was suitable for this study in the sense that previously established urban growth targets were achievable. There are, however, some legal constraints to growth that were not incorporated into this growth scenario. Limitations on land conversion such as wetlands protection, conservation easements, and federal and state lands would all prevent land use change to occur in certain areas. An improved approach to urban growth models would integrate cellular automata for simulating land use changes. A cellular automata generally works by simulating the present by extrapolating from the past using image time-series. It can also incorporate restrictions, such as those mentioned above, into the growth model. Recent studies have successfully used cellular automata to simulate a wide range of urban phenomena, including regional growth and urban sprawl (Hegde et al. 2008). It would be interesting to see future studies employing such techniques in valuation studies that try to model urban development.

In conclusion, while there is much room for improvement in terms of obtaining more accurate values for the ecosystem services provided by the Red Hills region, and for evaluating the changes in values resulting from land conversion, this study provides some good first order estimates. Additional GIS data would help with the analysis of historic land use patterns, and with future land use projections. And additional primary studies valuing services specific to the Red Hills region will help improve the estimates presented in this study. One such study is already being carried out. A cooperative research project between the University of Georgia and the Tall Timbers Research Station is currently underway, which will use stated preferences to value ecosystem services provided by the Red Hills region. Its results will go a long way towards obtaining estimates that are tailored to study site of this research.

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APPENDIX A

NLCD 2001 LAND COVER CLASS DEFINITIONS*

Open Water – All areas of open water, generally with less than 25% cover of vegetation or soil.

Urban Green – Includes areas with a mixture of some constructed materials, but mostly vegetation in the form of lawn grasses. Impervious surfaces account for less than 20 percent of total cover. These areas most commonly include large-lot single-family housing units, parks, golf courses, and vegetation planted in developed settings for recreation, erosion control, or aesthetic purposes

Urban Developed – Includes areas with a mixture of constructed materials and vegetation, such as single-family housing units (low intensity). Also includes highly developed areas where people reside and work in high numbers, such as apartment complexes, row houses and commercial/industrial (high intensity. Impervious surfaces account for 20-100 percent of total cover.

Barren Land – Barren areas of bedrock, desert pavement, scarps, talus, slides, volcanic material, glacial debris, sand dunes, strip mines, gravel pits and other accumulations of earthen material. Generally, vegetation accounts for less than 15% of total cover.

Deciduous Forest – Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. More than 75 percent of the tree species shed foliage simultaneously in response to seasonal change.

Evergreen Forest – Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. More than 75 percent of the tree species maintain their leaves all year. Canopy is never without green foliage.

Mixed Forest – Areas dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. Neither deciduous nor evergreen species are greater than 75 percent of total tree cover.

Shrub/Scrub – Areas dominated by shrubs; less than 5 meters tall with shrub canopy typically greater than 20% of total vegetation. This class includes true shrubs, young trees in an early successional stage or trees stunted from environmental conditions.

Grassland/Herbaceous – Areas dominated by grammanoid or herbaceous vegetation, generally greater than 80% of total vegetation. These areas are not subject to intensive management such as tilling, but can be utilized for grazing.

Pasture – Areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Pasture/hay vegetation accounts for greater than 20 percent of total vegetation.

Cropland – 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. Crop vegetation accounts for greater than 20 percent of total vegetation. This class also includes all land being actively tilled.

Wetland – Areas where forest or shrubland vegetation accounts for greater than 20 percent of vegetative cover and the soil or substrate is periodically saturated with or covered with water.

* Taken from NLCD 2001

APPENDIX B

DESCRIPTION OF ECOSYSTEM SERVICES FROM BENEFIT TRANSFER*

• ***Climate Regulation:*** Through photosynthesis, forest and other plant covers process and store carbon dioxide generated from humans and human activities, and convert it into the oxygen we all need to live and breathe. This capture-and-storage process is called carbon sequestration. Carbon sequestration by natural and agricultural land cover is estimated to process one-third of all human-generated CO2 emissions in a cost-effective way. The US Department of Energy has noted, "In the near-term, sequestration of carbon in terrestrial ecosystems offers a low-cost means of reducing net carbon emissions with significant ancillary benefits: restored natural environments for plants and wildlife, reduced runoff, and increased domestic production of agriculture and forest products."20 Forest cover also reduces reflected sunlight, which can raise atmospheric temperatures.

• ***Freshwater Regulation and Supply:*** Water is essential to life, and one of our most valuable natural assets. When local water supplies fail, water must be imported from elsewhere at great expense, must be more extensively treated (as in the case of low stream flows or well levels), or must be produced using more expensive means (such as desalinization). Forest and its underlying soil, and wetlands, play an important role in ensuring that rainwater is stored and released gradually, rather than allowed to immediately flow downstream as runoff.

• ***Waste Assimilation:*** Both forests and wetlands provide a natural buffer between human activities and water supplies, filtering out pathogens, nutrients such as nitrogen and phosphorous (often released as runoff from fertilizer and septic tanks), as well as metals and sediments. This "free" service provides benefits to humans in the form of clean drinking water, and sustains plants and animals by reducing harmful algal blooms, dissolved oxygen and excessive sediment in water. In an often-cited case, New York City concluded it was more cost-effective to preserve land around its water supply in the Catskill/Delaware watershed, and utilize its natural water filtration services, than to build a new water treatment plant.21 Trees also improve air quality by filtering out particulates and toxic compounds.

• ***Nutrient Regulation*:** The proper functioning of any ecosystem is dependent on the ability of plants and animals to utilize nutrients such as nitrogen, potassium and sulfur. For example, certain bacteria take nitrogen in the atmosphere and "fix" it such that it can be readily absorbed by the roots of plants. When plants die or are consumed by animals, nitrogen is "recycled" back into the atmosphere. Farmers apply tons of commercial fertilizers to croplands each year in part because this natural cycle has been disrupted by cultivation.

• ***Habitat Refugium*:** Contiguous "patches" of landscape with sufficient area to support naturally functioning ecosystems support a diversity of plant and animal life. As patch size decreases, and as patches of habitat become more isolated, population sizes, especially of rare plant and animal species, can decrease below the threshold needed to maintain genetic variance, withstand stochastic events and population oscillations, and meet social requirements such as breeding and migration. Large contiguous habitat blocks, such as forest or wetland, thus function as critical population sources for dispersing plant and animal species that humans value.

• ***Soil Retention and Formation:*** Soils provide many of the services mentioned above: water storage and filtering, waste assimilation, and a medium for plant growth. Natural systems both create and enrich soil, through weathering and decomposition, and reduce erosion.

• ***Disturbance Prevention:*** Natural wetlands and floodplains can help mitigate the effects of floods by containing stormwater. Wetland vegetation can reduce the damage of wave action and storm flows. The cost of floods in the US in terms of insurance claims and aid exceed $4 billion per year. Residents in Napa Valley recently concluded that it was better to let nature contain floods in a historic floodplain than build a dam, reaping recreational and tourism benefits as well as reducing losses caused by floods.

• ***Pollination:*** More than 218,000 of the world’s 250,000 flowering plants, including 80 percent of the world’s species of food plants, rely on pollinators for reproduction. Over 100,000 invertebrate species—such as bees, moths, butterflies, beetles, and flies—serve as pollinators worldwide. At least 1,035 species of vertebrates, including birds, mammals, and reptiles, also pollinate many plant species. At least 50 pollinator species are listed as threatened or endangered by the US Fish and Wildlife Service. Even the common honeybee has been impacted by loss of habitat and indiscriminate use of pesticides, with wild honeybee populations down 25 percent since 1990.

• ***Recreation and Aesthetics:*** Intact natural ecosystems attract people who fish, hunt, hike, canoe, or kayak, bringing direct economic benefits to the areas surrounding those natural areas. People's willingness to pay for local meals and lodging, and to travel, are economic indicators of the value they place on natural areas. Real estate values, and therefore property-tax revenues, often increase when a house is located near protected open space. The difference in real estate value reflects people's willingness to pay for the aesthetic and recreational attributes of open space.

*These descriptions are taken from the technical notes of Wilson and Troy's 2003 study entitled "Accounting for the economic value of ecosystem services in Massachusetts."

APPENDIX C

LAND COVER DISTRIBUTION BY DATASET

	National Land Cover Data (NLCD)				Coastal Change Analysis Program (C-CAP) Land Cover Data					
	1992		2001		1996		2001		2006	
	ha	*%*	*ha*	*%*	*ha*	*%*	*ha*	*%*	*ha*	*%*
Open Water	5002	1.79%	1565	0.56%	2330	0.83%	1751	0.62%	1780	0.63%
Urban Green Space	565	0.20%	22103	7.89%	8204	2.92%	8385	2.99%	8545	3.04%
Urban Developed	20391	7.28%	7550	2.70%	9627	3.43%	9827	3.50%	10124	3.61%
Barren Land	33346	11.90%	134	0.05%	203	0.07%	250	0.09%	346	0.12%
Forest	114851	41.00%	150150	53.60%	120673	42.99%	123156	43.87%	118964	42.38%
Shrub/Scrub	599	0.21%	718	0.26%	19698	7.02%	20522	7.31%	22246	7.93%
Grass/Herbaceous	10525	3.76%	26028	9.29%	19160	6.83%	15783	5.62%	17826	6.35%
Pasture	6126	2.19%	12987	4.64%	13328	4.75%	13455	4.79%	13405	4.78%
Cropland	38785	13.85%	14000	5.00%	14754	5.26%	15032	5.36%	14979	5.34%
Wetland	49937	17.83%	44894	16.03%	72723	25.91%	72537	25.84%	72485	25.82%