The Application of Ecosystem Service Markets to the Conservation of Red Wolf Habitat in North Carolina

A Local Effort with National Implications

Courtesy U.S. Fish and Wildlife Service

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DEFENDERS OF WILDLIFE

NICHOLAS INSTITUTE FOR ENVIRONMENTAL POLICY SOLUTIONS
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Executive Summary

Red wolves that were once extinct in the wild were reintroduced to Eastern North Carolina wildlife refuges, and over a hundred wolves have since spread out across five counties. In spite of interest in supporting habitat favorable to the red wolf re-establishment, conservation programs currently available to farmers do not alter cropland use patterns in these five counties enough to substantially benefit the wolves. These programs are designed to idle cropland to address problems such as erosion on highly erodible land and loss of wetlands, which occur on a relatively small portion of U.S. cropland area. Only one option considered, carbon sequestration payments for forest establishment, could alter the Eastern North Carolina landscape sufficiently to substantially support red wolf establishment. However, programs currently do not offer large enough payments to encourage carbon sequestration practices.

Efforts to bring the U.S. into international carbon trading programs to address global warming have not yet succeeded. Had cap and trade programs for carbon been developed, economic analysis suggests red wolves as potential beneficiaries. International carbon prices in the $30/ton CO2e range likely would be sufficient to induce area farmers to plant forests on their cropland in exchange for carbon sequestration payments. Such payments might compete especially with modestly productive farming activity that occurs in portions of Eastern North Carolina. Shifts from cropland use to forest plantations likely would take place on a scale to substantially alter the local landscapes and support forest habitat preferred by red wolves.

In the absence of the establishment of international carbon trading in the U.S., carbon prices might continue in the $5/ton CO2e range. At this price, planting trees to sequester carbon does not compete with crop production, so it has little chance of generating significant shifts of cropland to forest uses.

Some activities related to forests potentially produce large benefits, but for these activities, no effective options are currently available. For example, existing forests in the four counties capture large amounts of carbon, but these carbon sequestration benefits are lost when the forests are harvested. Increased forest area also offers open space amenities that increase the value of homeowners' property by about $5,000/property, as local residents enjoy the open space. However, no markets exist to exploit these opportunities to benefit home owners.

Forest plantation can offer many benefits in addition to its reduction of greenhouse gases. For example, replacing some cropland with forests would greatly reduce nutrient run-off into regionally and nationally important water bodies, such as the Pamlico Sound. Included in this list of potential benefits is the support that forest habitat would offer to red wolf establishment.

I. Introduction

There is a growing recognition of the importance of a variety of useful services provided by healthy ecosystems, termed ecosystem services. These include air and water purification, flood control, climate regulation, and plant pollination. This has spawned research efforts and policy initiatives to encourage better management of ecosystems to enhance the flow of these services (Heal, 2000). Since much of the production of ecosystem services occurs on private lands, it is particularly important to understand how landowners and operators can be incentivized to enhance ecosystem service flows (Kroeger and Casey, 2007). Furthermore, a better understanding of the often undervalued benefits of ecosystem services and how those services can be marketed can contribute toward improved ecosystem health as well as improved economic well-being for land managers (Kramer, 2007).

This project examines the ecosystem service benefits associated with conserved red wolf habitat. After being declared extinct in the wild in 1980, red wolves were reintroduced in 1987 by the U.S. Fish and Wildlife Service into the Alligator River National Wildlife Refuge on North Carolina's Albemarle Peninsula (US FWS, 2007). The wild population of red wolves is currently estimated at 100 to 120 individuals spread across five eastern North Carolina counties (US FWS, 2010). These 5 counties (Hyde, Dare, Tyrell, Beaufort, and Washington) constitute the spatial extent of the Federal government's Red Wolf Recovery Program (RWRP) (see Figure 2.1). As social animals, the wolves are clustered in some twenty packs distributed over 1.7 million acres (US FWS, 2010). While originally released on public land, many of the wolves spend at least part of their time on private forest and farm land in the area.

This report describes Phase 2 of a project on the use of market-type incentives to encourage the conservation of red wolf habitat in eastern North Carolina. An earlier report on Phase 1 summarized landowners' knowledge and attitudes towards payments for ecosystem services, including payments for conserving and maintaining red wolf habitat (Kramer and Jenkins, 2009). A survey was administered by mail to a random sample of 298 farm operators in the red wolf area about their attitudes toward current conservation programs and their interest in participating in future programs that are oriented toward the provision of ecosystem services. The survey was implemented in the five RWRP counties and in adjoining Bertie County. The findings showed that approximately one-half of farm operators in the study area have participated in conservation payment programs in the past and that they are generally satisfied with their participation experience (Appendix 1, Table A1.2). While there is a lack of familiarity with ecosystem services terminology, many are interested in participating in future payment for ecosystem service (PES) programs, particularly if the programs emphasize wildlife conservation or water quality. Payment levels are found to be an important factor in decisions to enroll, but so are other program attributes, particularly contract length and program administration type (Appendix 1, Table A1.2). The survey data indicates that a PES that is specific to red wolf conservation is not supported by most farmers. The results also show that targeted marketing and information campaign could be used to address a lack of familiarity with ecosystem services and markets and promote future sign-ups.

Phase II builds upon Phase I and consists of the following tasks: (1) quantifying ecosystem service benefits associated with conserving and restoring habitat with a focus on carbon storage; (2) conducting a cash flow analysis of the potential contribution of payments for ecosystem services to farms representative of the study area; and (3) estimating ecosystem service benefits to home owners associated with providing the above open space**.**

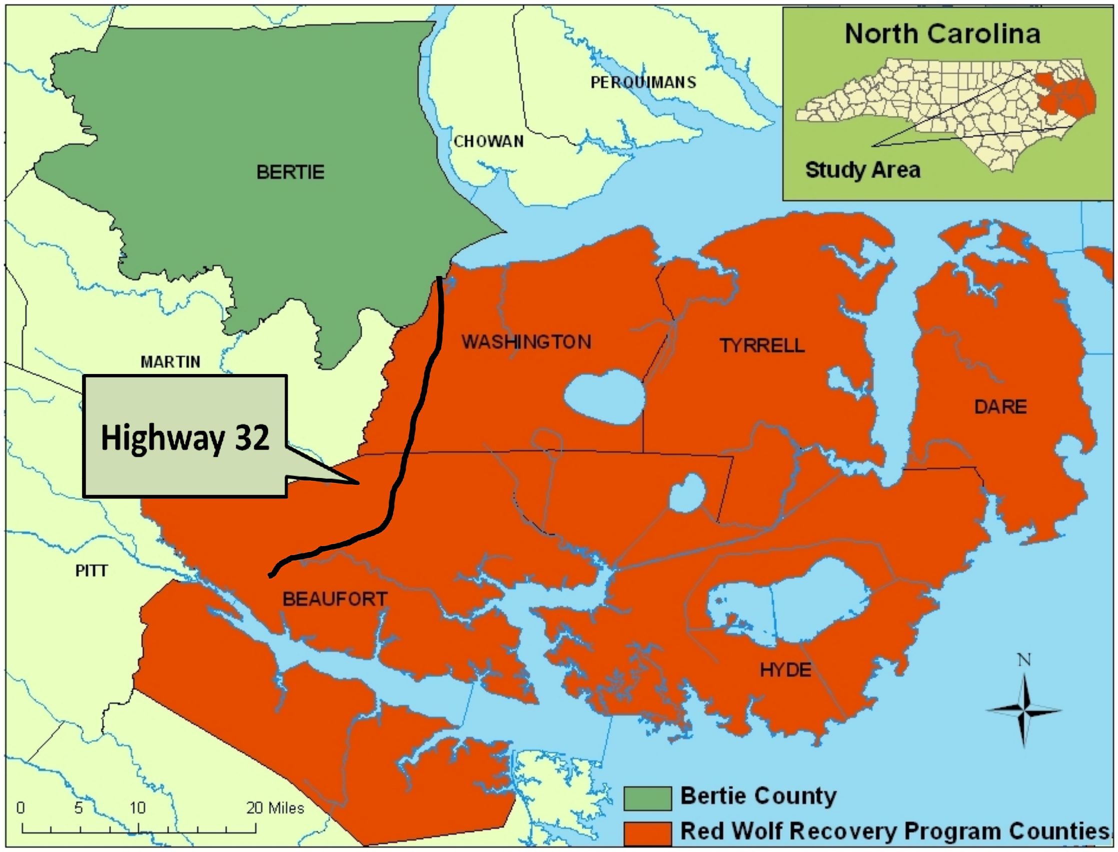


Figure 2.1: Map of Study Area

II. On-Farm Ecosystem and Financial Benefits from Carbon Sequestration on Habitat Lands

This research is particularly timely because the 2008 Farm Bill takes a first step towards encouraging farm operator participation in emerging markets for ecosystem services. Guidelines are being developed at the USDA to inform new ways to provide payments for ecosystem services such as carbon sequestration. The research described below will help inform this process. Our study also considers the contribution of traditional farm conservation programs, e.g. Conservation Reserve Program, to farm income in the study area. These conservation programs can be considered an early form of PES, although unlike the newly emerging PES programs, traditional agricultural conservation payments have been tied to particular land use activities rather than to the provision of a particular service (Ferraro, 2001). Methodologically, the study follows the tradition of farm level cash flow analysis of conservation incentives combining economic and ecological data (Bosch et al., 2008; Goldstein et al., 2006, Shyamsundar and Kramer, 1987).

Carbon storage estimates

When land is conserved as red wolf habitat, it provides direct benefits, but it can also generate other ecosystem services. One of the more promising opportunities for landowners to derive market income from these services is the emergence of carbon markets. Carbon offset payments could provide financial incentives to change tillage practices and to engage in afforestation projects to plant trees on cropland. These measures could also potentially increase the extent of red wolf habitat.

Estimates of potential carbon storage flows for afforested agricultural lands are taken from the US Forest Service 1605(b) tables generated by the FORCARB model for various forest types in various regions in the U.S. (Smith et al., 2006). We use the estimates for the Southeast region and the dominant natural forest types the two zones in the study area.

One representative Farm A (described below) in the coastal plain, we assume that the owner afforests the longleaf-slash pine forest type. As the trees grow, they sequester carbon in live biomass, while carbon also accumulates in the litter, understory, and soil under the trees. Gains in carbon range from 2.9 to 3.8 tCO_2/yr over the 40-yr study period. On representative Farm B in the tidewater area, we assume that the owner afforests the oak-cypress-gum forest type. New carbon sequestered in the various carbon pools ranges from 2.8 to 4.8 tCO_2/year over 40 years Tables A2.1 and A2.2 in Appendix 2 present the carbon growth curves for the two relevant forest types.

In addition to afforestation, landowners in the study area could switch to no-till production methods to increase soil carbon levels. For North Carolina, Chicago Climate Exchange (CCX) uses the standard rate of 0.6 tonnes CO_2 sequestered per year per acre of crop field employing minimal tillage practices. Each of the crop types we model has a no till or strip till option, both of which are accepted planting methods for continuous conservation tillage under the CCX Soil Sequestration Offset Project Protocol (Chicago Climate Exchange, 2009). In the Carbon scenarios, all crops except peanuts are switched from conventional to conservation tillage. According to the NCSU crop budgets, there is not a reduction in productivity associated with going no till and, in all cases, production costs are somewhat lower under no till (1-7% lower per acre). The overall carbon benefits of changing to conservation tillage are likely to be underestimated here since we do not account for the emissions avoided resulting from lower usage of farm machinery and reduced application of fertilizers.

Cash flow modeling

Based on the results of the carbon storage analysis above, and with estimated financial flows from available conservation payment programs, we conducted cash flow analysis for two representative farms that could provide red wolf habitat on the Albemarle Peninsula and on the adjoining mainland. This analysis compares conventional agricultural and forestry market returns to those available from diversified operations that could include ecosystem service payments for carbon storage. This cash flow analysis builds upon previous research at Duke University that conducted cash flow analyses for representative NC properties in the mountain, Piedmont, and coastal plain ecoregions to determine income potential from carbon storage and existing conservation programs (Gray, 2008).

The starting point for the cash flow modeling was the identification and description of representative farms. Overall, the land use patterns of the study area consist of a mix of cropping and forestry activities (Appendix 1, Tables A1.3 and A1.4). The predominant crops are commodity crops, such as soybeans, cotton and corn (Appendix 1, Table A1.5). Participation in conservation programs tends to be lower than in some other parts of the state, though 51% of survey respondents report participating at some point. Highway 32, running north and south, in western Washington and Beaufort Counties splits the area into two agricultural zones— coastal plain to the west and tidewater to the east. Soils west of highway 32 are sandier, so there is more peanut and cotton production. Generally, the farms to the east are larger with proportionally more acreage in crops. Based on our survey data, agricultural census data, and discussions with area agricultural experts, two representative farms were specified (see Figure 2.2 and Appendix 1). Farm A, representing farms west of highway 32, consists of 500 acres, with 60% in field crops and 25% in planted forest for timber. Representative of farms that are located east of highway 32, Farm B is 1000 acres, with 75% used for crop production and 12.5% in planted forest. Although little of the RWRP counties area falls west of Highway 32 (see Figure 2.1), we included a representative farm from that area under the assumption that red wolves might continue their westward expansion and that incentivizing the increase in habitat in that area would be a pertinent issue in the future.

Farm Attributes	FARM A	FARM B
Location	West of Hwy 32	East of Hwy 32
Size	500 acres	1000 acres
Land use mix: **Crops** **Planted forest** **Natural forest**	 300 acres (60%) 125 acres (25%) 75 acres (15%)	 750 acres (75.0%) 125 acres (12.5%) 125 acres (12.5%)
Crop mix	Corn 30% Cotton 25% Soy 15% Peanuts 30%	Corn 30% Cotton 20% Soy 30% Wheat 20%

Figure 2.2: Representative Farms, business-as-usual (BAU)

Cash flow models were constructed in MS Excel for each representative farm. Crop budgets for the five relevant crop types in the two agricultural zones were obtained from North Carolina State University (NCSU) Agricultural and Resource Economics Department (NCSU, 2010). Crop revenues and costs are averaged over a three-year period from 2008-2010 to reduce the effects of volatile input and crop prices. Timber data were obtained from NCSU Forestry Extension. Conservation program data were obtained from Natural Resource Conservation Service (NRCS) and North Carolina Department of Environment and Natural Resources (NC DENR) to estimate the potential income from conservation program enrollment. Potential carbon payment levels were calculated based on prices per tonne CO_2e[1] in the emerging markets for carbon that may be accessed by landowners. The low carbon price, \$5/tCO_2e, is reflective of the range of prices available in the voluntary OTC market in 2008[2] (Hamilton et al., 2009). The high carbon price, \$30/tCO_2e, assumes the establishment of a compliance-based cap-and-trade system with an accompanying offset program allowing forest and agricultural projects. Updated EPA analysis of the Waxman-Markey (H.R. 2454) climate change bill indicated that carbon prices could be on the order of \$32-3/tCO_2e in 2030 over a time period of 2012-2050 (US EPA, 2010).

For each representative farm, five scenarios are modeled, yielding the net present value (NPV) of returns from agricultural, forestry, conservation, or carbon activities over a 40-year time horizon (years 2010 to 2050). We use a 6% discount rate to discount future cash flows

[1] The recognized carbon commodity traded in markets is one tonne CO_2e, meaning a metric ton (1000 kilograms) of carbon dioxide equivalents. Greenhouse gases (e.g., N_2O, CH_4) have differing global warming effects per unit; in order to standardize to one emissions currency, these are all expressed as the number of CO_2 molecules necessary to cause that effect.

[2] For over-the-counter (OTC) carbon offset trades in 2008, volume-weighted average credit prices were \$3.35/tCO2e and \$7.50/tCO2e for credits generated from agricultural soil sequestration and afforestation/reforestation conservation respectively.

back to the present. The baseline or business-as-usual (BAU) scenario reflects the conventional management of farms in the two-part study area, as detailed in Figure 2.2. This includes harvesting of the planted forest at the end of the 40-year study period, as well as costs and benefits of forest management during the period. Likewise, annual hunting lease revenues are integrated into the baseline. The net returns generated from the baseline were then compared to those from two scenarios that included conservation program enrollment and from two other scenarios that included afforestation and no-till activities to generate carbon payments. These alternative scenarios reflected carbon or conservation payments net of the costs of switching land use activities. The land use mixes for each scenario are shown in Figures 2.3 and 2.4 for the two representative farms.

The low conservation scenario is formulated by placing 15% of cropland into the Conservation Reserve Program (CRP) and by switching all of the soybean production into no-till. NC Agricultural Cost Share program payments compensate the farm operator $25/acre/year for moving into no-till, though the lifetime cap of $25,000 limits the amount that can be received over the 40-year time horizon. Under the high conservation scenario, an additional 15% of cropland is removed from crop production and enrolled in the Wetland Reserve Program (WRP), which gives landowners an up-front, lump-sum payment for restoring a wetland and placing a permanent easement on the selected acreage[3]. The low and high carbon scenarios share the same land use composition and only differ by the assumed carbon price ($5 or $30/t$CO_2$e). Positing the establishment of a carbon market, farm operators gain incentive to consider "growing carbon" as a viable alternative to traditional commodity production. In response, 15% of cropland is afforested with the local dominant native forest type and all crop production is changed to no-till or strip-till, except for peanuts. Carbon sequestered through these measures is calculated annually and carbon credits are sold. From the revenue earned from credit sales, we subtract out carbon registry and project aggregator fees as well as 10% of credits that are not returned from the 20% required to be placed in the buffer reserve. Note that when land transitions out of crop production under the Conservation and Carbon scenarios, the acreage comes out of the crop type(s) with the lowest profitability. The lowest margin crop is cotton for Farm A and wheat for Farm B.

We assume that carbon prices stay constant throughout the 40-yr study period. In some larger modeling efforts, it is common to have carbon prices grow at 5% a year, to emulate the carbon price trajectory from EPA analyses, or to leave carbon prices constant. We have left them constant because our analysis also involves agricultural commodity, fertilizer, and timber prices, which we also leave uninflated over time. Additionally, we assume there is minimal development pressure on agricultural lands in the RW study area. Thus, we focus on variable costs of the farming operations and do not include land values. The one exception to this is that we do include the fixed costs of farm machinery.

[3] Personal communication with Mike Hinton, North Carolina NRCS. Per acre payments for good quality agricultural land are about $3,200 for the coastal plain and about $2,700 for the tidewater zone.

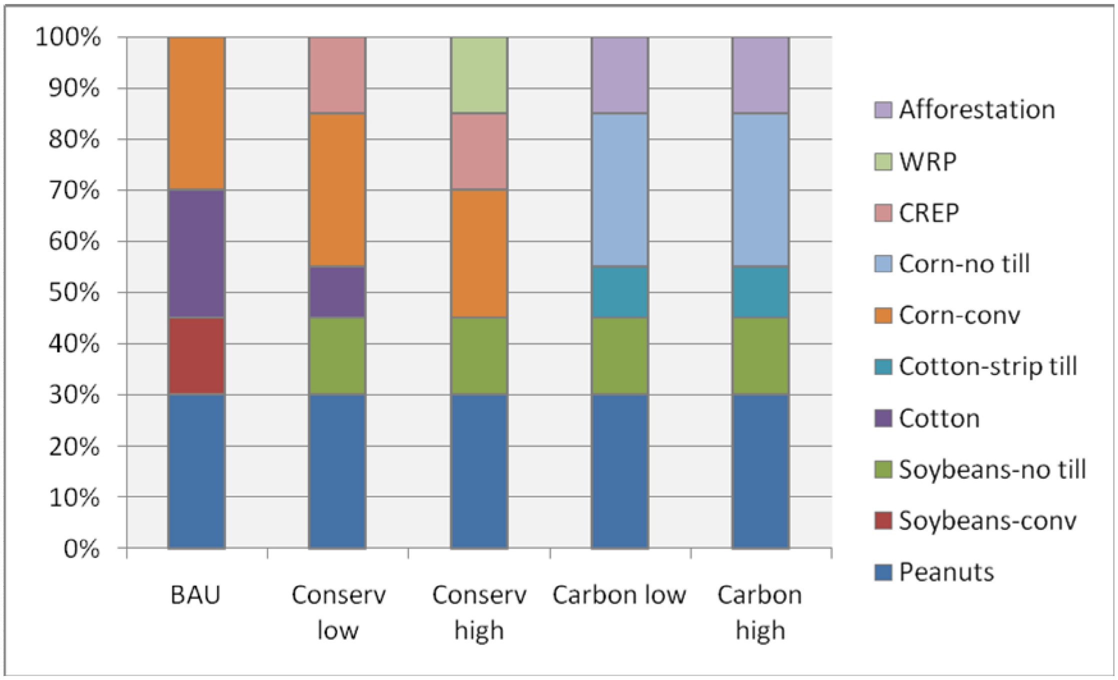


Figure 2.3: Proportional use of cropland under five scenarios for Farm A

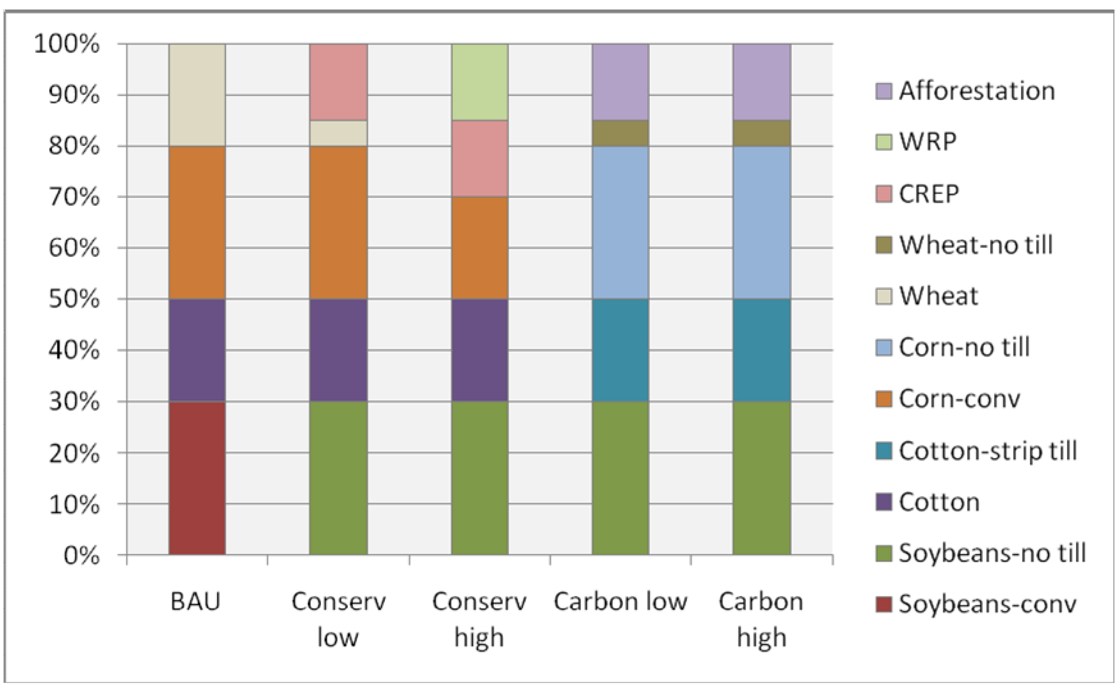


Figure 2.4: Proportional use of cropland under five scenarios for Farm B

In our analysis, we treat the conservation and carbon scenarios as mutually exclusive. This is because it is likely there will be a legal requirement of additionality to earn carbon credits that will not allow carbon sequestered due to practices driven by conservation payment programs to be eligible for carbon payments.

Results

Results from the cash flow modeling are shown in Figure 2.5. In addition to the baseline for each farm, results are presented for a low and high conservation scenario and for a low and high carbon scenario.

The baseline (BAU) NPV is $1486 per acre for Farm A and $1653 for Farm B. Under the low conservation scenario, there is little change in income since the CRP payments are mostly offset by the lost crop income from taking land out of production. However, under the high conservation scenario, NPV increases by $193 per acre (13%) on Farm A and almost 8% on Farm B. This boost owes to the high, upfront (undiscounted) payments from WRP. These payments are higher for coastal plain farmland and thus benefit Farm A more than Farm B.

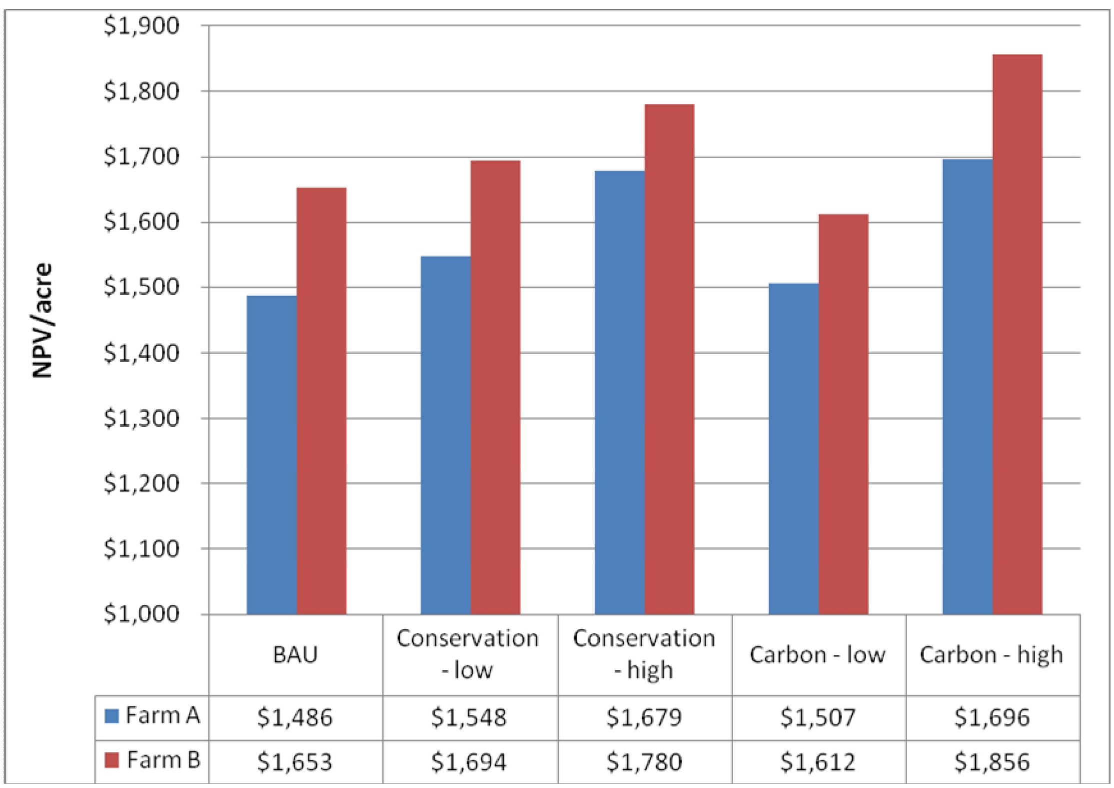


Figure 2.5: NPV of Representative Farm Returns[4]

[4] Note that the y-axis begins at $1,000 NPV/acre.

Next we consider the carbon payment scenarios. A low carbon price of $5 per tonne CO_2e means lower returns than the baseline for Farm B and basically equivalent returns for Farm A. At this low price, afforesting 15% of one's cropland earns less per acre than the crops previously planted. There is some additional income from soil carbon-generated credits from no-till, but not enough to offset the opportunity cost of taking some land out of production to plant trees. For each farm, the high carbon price scenario ($30/t$CO_2e$) generates the highest NPV/acre out of all the scenarios. It represents 14% and 12% gains above the baseline discounted returns for Farm A and B respectively. For Farm A, the returns in the high carbon scenario are comparable to the scenario that emphasizes conservation program enrollment (CRP, NC Ag Cost Share, and WRP), while high carbon is clearly the most attractive scenario for Farm B.

Overall, the impacts of conservation program enrollment and carbon market participation are relatively similar across the two farms, though few farms would shift to "grow carbon" if the carbon price were relatively low. From an income perspective, although the high conservation and high carbon market scenarios dominate the other options, other factors may also be important to landowners' land management decision process.

Discussion

Incentives for ecosystem service provision offer potential opportunities for farmers to capture the value over time of ecosystem services they provide through their conservation activities. Our cash flow modeling provides a window into the farm level economics of ecosystem service payments in the two agricultural zones ineastern North Carolina. To the extent that the representative farms reflect the economic environment of individual farm operators, the results suggest that those who are currently choosing not to participate in government run conservation programs are foregoing some income as compared to the high conservation scenario we analyzed. Why would farmers not avail themselves of existing income opportunities from CRP, WRP and NC Agricultural Cost-Share programs? Our survey (Phase I) found that the most commonly cited reason for not participating in conservation programs is "concern about government restrictions on private property" (Appendix 1, Figure A1.1). Other respondents said they did not want to change their land management practices, did not want to deal with program paperwork and requirements, or that payment levels were not high enough. Thus, while our results suggest that it could be profitable for some land managers to enroll in a mixture of conservation programs, there are many understandable explanations for the current low level of involvement (33%) in the publicly funded PES programs.

In addition, carbon payments would appear to be a good option for land managers looking to increase or diversify their income. The results show that the representative farms could put substantial portions of their land into carbon storing activities that could improve wildlife habitat, while raising income over the planning horizon. However, this scenario is contingent on the establishment of a cap and trade policy on carbon emissions that would generate a price for carbon in the range of $30/t$CO_2e$. The same conservation activities would lead to a lower than baseline level of returns if the current level of carbon prices given by voluntary markets persist.

Note that a major difference between the no-till practiced under the Conservation scenarios and under the Carbon scenarios is that the former is mediated by a government program (NC Agricultural Cost Share), while the latter is market-driven. The payments in the first case are limited by a lifetime cap set by the program; in the second case, farm operators' interest in conservation tillage would depend on carbon offset price levels.

Future research could explore the relative risk of different land use activities. Conservation program payments are established by legislated government programs and are likely to be less volatile than payments from future carbon markets. Hence, it is would be useful to apply risk analysis to consider how price uncertainty might affect PES decision making. Also, given the public interest in maintaining the quality of river and coastal waters, future research could examine the cash flow impacts of an ecosystem service market for nutrient reductions. Land use changes that enhance wildlife habitat, such as buffer strips along streams, could also reduce farm runoff, thereby enhancing water quality and generating credits for a future nutrient trading market.

III. Natural Lands Carbon Sequestration

Our focus in this section is on undeveloped lands, that is, areas that have not been converted to crop production or for residential, commercial, industrial or infrastructure uses. We include plantations and managed forests in our analysis because these systems under certain conditions can be net carbon sinks that can store carbon for long periods of time in the form of long-lived wood products (Smith et al., 2006).

Plants take up carbon dioxide (CO_2) from the atmosphere during the process of photosynthesis. A portion of the sequestered carbon is returned to the atmosphere through plant respiration and decomposition of organic matter, but some is retained for long period of time in the form of new live plant tissues and organic matter accumulations in soils and sediments. During the recent centuries and especially decades of increasing anthropogenic carbon emissions, undeveloped lands as a whole have been acting as a carbon sink (Canadell et al., 2007). However, the sign and size of the net flux differs among different plant species and vegetation communities, and it differs for the same species or community at different locations.

To estimate the net carbon balance of undeveloped lands in the study area, we first identified the total acreage of different vegetation community types using National Land Cover Data (NLCD) information (Table 3.1). We then used more detailed information on specific vegetation community types and species composition (North Carolina Wildlife Resources Commission, 2005) to match the vegetation types found in the study area to net carbon sequestration estimates reported in the literature (Table 3.2). To reduce the impact of climatic factors on the applicability of the literature estimates to our study area, we focused on studies conducted in the Southeast.[5] The exceptions are emergent wetlands for which we were unable to locate studies in the region.

Table 3.1: Land cover of non-developed and non-agricultural lands in study area

Land cover	*Beaufort*	*Dare*	*Hyde*	*Tyrell*	*Washington*	*Total area*
Deciduous Forest	3,618	286	3,622	4,572	10,303	22,402
Coniferous Forest	41,234	6,270	38,262	30,069	37,153	152,987
Mixed Forest	2,220	1,642	3,633	2,530	4,152	14,177
Shrubland/scrubland	8,903	4,780	24,310	7,541	14,458	59,993
Grassland	12,669	6,940	27,592	11,731	18,683	77,615
Woody Wetlands	25,969	214,033	232,023	187,304	78,897	738,225
Emergent Herbaceous Wetlands	3,304	44,861	80,792	6,230	3,355	138,542

Source: Based on analysis of NLCD 2001 GIS data.

5 The region of the U.S. in which a forest is located (based on the broad regions identified in Smith et al., 2006) is a significant factor influencing forest ecosystem carbon, in addition to forest type, previous land use, management and productivity (Smith et al., 2006).

Net carbon balances for conifer and hardwood forests, various types of wetlands, grasslands and shrub lands are reported in Table 3.2. These balances represent the results of various studies conducted in the southeastern United States for stands that have not been experimentally carbon-fertilized, or adjusted for differences in the lengths of the growing season between the literature sites and our study site. The sequestration estimates take into account carbon dioxide (CO_2) flows only.

Table 3.2: Net carbon balances for ecosystem types found in the study area

Ecosystem type	*C flux measured*	*Net C balance*	*Location*	*Source*	*Annual Mean Temp.*	*GSL-adjusted NEE*
		tC/ac/yr			*°C*	*tC/ac/yr*
CONIFER FOREST/PLANTATION						
Even-aged loblolly pine plantation; median height 13m, mean dbh 15 cm	NEE	3.801	Orange Co., NC	Hui et al (2003)	15.6	4.014
15-yr-old loblolly pine plantation (Hui et al. 2003)	NEE	1.733	Orange Co., NC	Hamilton et al. (2002)	15.6	1.830
17-yr old loblolly pine plantation	NEE	2.587	Orange Co., NC	Lai et al. (2002)	15.6	2.587
11-yr-old (young) loblolly pine plantation *	NEE	-0.445	Laurinburg, NC	Maier & Kress (2000)	16.7	-0.432
Slash pine plantation, dominated by 24 yr-old trees	NEE	2.729	Alachua Co., FL	Clark et al. (1999)	20.6	1.790
Slash pine plantation clearcut (0-2 years old)	NEE	-4.354	Alachua Co., FL	Clark et al. (2004)	20.6	-2.856
Slash pine plantation, 10-yr old	NEE	2.387	Alachua Co., FL	Clark et al. (2004)	20.6	1.566
22-yr old (in 2005) planted *Pinus taeda* forest:						
STE method	NEE	1.498	Orange Co., NC	Stoy et al. (2006)	15.6	1.582
NRH method	NEE	1.838	Orange Co., NC	Stoy et al. (2006)	15.6	1.838
60-yr old, naturally regenerated longleaf/slash pine stand	NEE	0.721	Alachua Co. FL	Powell (2002)	20.6	0.473
Pine plantation	NEE	2.532	Orange Co., NC	Falge et al. (2001)	15.6	2.674
Maturing loblolly pine plantation	NEE	1.638	Orange Co., NC	Juang et al (2006)	15.6	1.730
HARDWOOD FOREST						
Southern hardwood forest	TAB+CR	2.125	Oak Ridge, TN	Greco & Baldocchi (1996)	15.0	2.346
80–100-year-old mixed deciduous forest dominated by oak and hickory species; STE method	NEE	1.126	Orange Co., NC	Stoy et al. (2006)	15.6	1.189
NRH method	NEE	1.798	Orange Co., NC	Stoy et al. (2006)	15.6	1.898
>70 yrs old upland oak forest; biometric low estimate	NEE	0.757	Oak Ridge, TN	Hanson et al. (2004)	15.0	0.836
>70 yrs old upland oak forest; biometric high estimate	NEE	1.008	Oak Ridge, TN	Hanson et al. (2004)	15.0	1.113
>70 yrs old upland oak forest; Eddy covariance-based estimate **	NEE	2.623	Oak Ridge, TN	Hanson et al. (2004)	15.0	2.896
WETLANDS - WOODY						
Pine-spruce wetland ***	TAB+SOC	1.724	29° N, FL	Li et al. (2004)	20.6	1.131
Uneven-aged mature unmanaged cypress wetland, oldest trees ~30 yrs ****	NEE	0.245	Alachua Co., FL	Clark et al. (1999)	20.6	0.161
Uneven-aged mature unmanaged cypress wetland, oldest trees ~30 yrs ****	NEE+FLD	1.083	Alachua Co., FL	Clark et al. (1999)	20.6	0.710

- over -

- Continued -

Ecosystem type	*C flux measured*	*Net C balance*	*Location*	*Source*	*Annual Mean*	*GSL-adjusted NEE*
		tC/ac/yr			*Temp. °C*	*tC/ac/yr*
WETLANDS - POCOSIN	TAB	0.064	Mid-Atlantic	FWS (2009)	n.a.	n.a.
WETLANDS - FRESHWATER MINERAL SOIL	LSS net C seq.	0.069	North America [$]	Bridgham et al. (2006)	n.a.	n.a.
WETLANDS - EMERGENT	NEE	3.55	Columbus,OH [#]	Altor and Mitsch (2008)	n.a.	n.a.
	NEE	8.51	Columbus,OH [##]	Altor and Mitsch (2008)	n.a.	n.a.
	NEE	0.74	Nueces Co., TX	Heinsch et al. (2004)	n.a.	n.a.
WETLANDS – TIDAL FRESHWATER MARSH	TAB	0.41	Altamaha River, GA	Craft et al. (2006)	n.a.	n.a.
GRASSLANDS					n.a.	n.a.
Old field; drought conditions in measurement year	NEE	-0.393	Orange Co., NC	Novick et al. (2004)	n.a.	n.a.
modeled CO_2 net balance under normal conditions	NEE	0.263	Orange Co., NC	Novick et al. (2004)	n.a.	n.a.
Old field, mowed at least once annually for forage:					n.a.	n.a.
STE method	NEE	-0.154[§]	Orange Co., NC	Stoy et al. (2006)		
NRH method	NEE	0.696[§]	Orange Co., NC	Stoy et al. (2006)	n.a.	n.a.
SHRUBLAND/SCRUBLAND	NEE	1.290	Brevard Co., FL	Powell et al. (2006)	22.4	0.660

Notes: Negative values indicate a net carbon source; positive, a net carbon sink. C – carbon; CR – coarse roots; FLD – fine litter deposition; GSL – growing season length; LSS - Landscape-scale sediment; NEE – net ecosystem exchange; NRH – non-rectangular hyperbolic; SOC – soil organic carbon; STE – short-term exponential fits; TAB – Total aboveground biomass. GSL adjustment based on White et al. (1999).

* Authors note that pine plantations are carbon sources during the first years; sink status begins later at higher latitudes and under drought conditions (see also Clark et al., 2004).

** Authors note that eddy covariance estimates for the site are expected to result in overestimates of NEE due to site-specific factors.

*** Negative GWP due to CH_4 emissions, unlike the Minnesota wetland also studied.

**** CH_4 emissions not included in analysis.

[$] Average

[#] Least productive of four created wetland

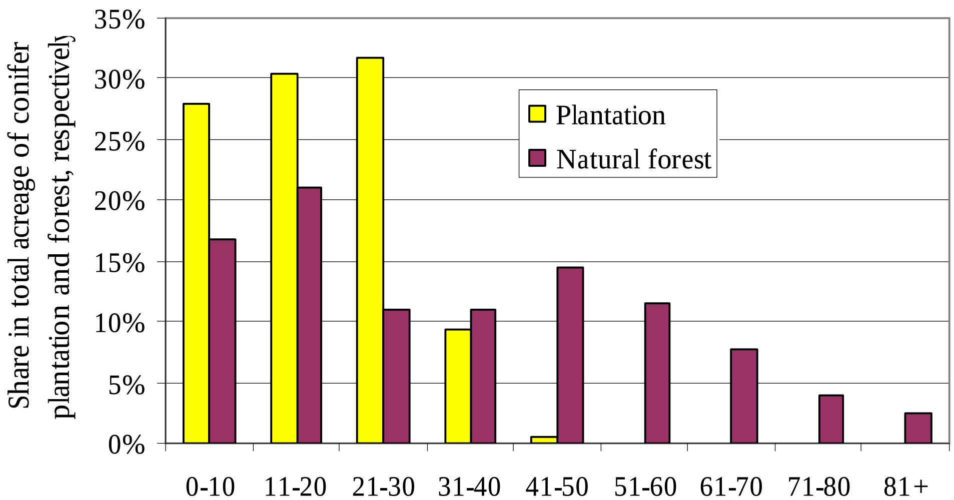
[##] Most productive of four created wetlands.

[§] Includes biomass removed during mowing.

Coniferous Forest

In the study area, this land cover type is represented by dry coniferous woodland dominated by slash and loblolly pine. This land cover includes many plantations (North Carolina Wildlife Resources Commission, 2005).[6] Most of these lands are privately owned and contain conifers at various maturity stages. Most carbon sequestration estimates in the literature for conifers in the southern U.S. are for young (0-11 years) and mature (22+ years) plantations, respectively, with few estimates for intermediate-age plantations (Table 3.2). In addition, we were only able to locate one estimate for the net carbon balance of a natural conifer forest.

The literature shows that pine plantations act as net carbon sources during the first years before becoming carbon sinks during later years (Sampson et al., 2008), with the timing of the switch occurring later at higher latitudes and under drought conditions (Clark et al., 2004; Maier and Kress, 2000). Because of this change in the net carbon balance of conifers with stand age, we use the observations from Table 2.2 to estimate a function that describes the net ecosystem exchange (NEE) of carbon for conifers as a result of tree age. We then use this function to develop estimates of the NEE of carbon for the wide range of stand age classes found in conifer plantations and natural forests in the study area (Figure 3.1).



Source: Conner (2003)

Figure 3.1: Age class composition of conifer plantations and natural forests in North Carolina's Northern Coastal Plain, 2000 [7]

Net carbon sequestration (NEE) (Figure 3.2) is a function of the particular growing conditions (precipitation, growing season length, temperature, soil fertility, disturbance events). Although we adjust the reported NEE estimates to the growing season length (GSL) in our study area (Table 3.2), it is beyond the scope of this analysis to attempt to account for differences in other growing

[6] Seventy-four percent of all pine plantations in the State occurred in the Coastal Plain, where 18 percent of the timberland was in pine plantations (Brown et al., 2006)

[7] The average rotation age in coastal NC loblolly pine plantations is 20-35 years (North Carolina Forest Service, 2006).

conditions and their impacts on NEE. For this reason, and because of the small number of observations, the estimated function that expresses NEE of carbon as a result of tree age is only an approximation.

The GSL adjustment of NEE values is based on White et al.'s (1999) findings that a 1°C increase in mean annual air temperature increases the length of the growing season for deciduous forests by approximately five days (an observed linear relationship over the entire range of mean annual temperature investigated, from 7-19 °C). A 1-day increase in GSL on average caused a 1.6% increase in net ecosystem production (NEP; equivalent to NEE [Hamilton et al., 2002]). We follow Diem et al. (2006) in assuming that NEE values for coniferous forests increase at the same rate and use the difference in mean annual temperatures between the locations studied in the literature (Table 3.2.) and our study area to perform the NEE adjustments. The GSL-adjusted NEE values are reported in the last column in Table 3.2.

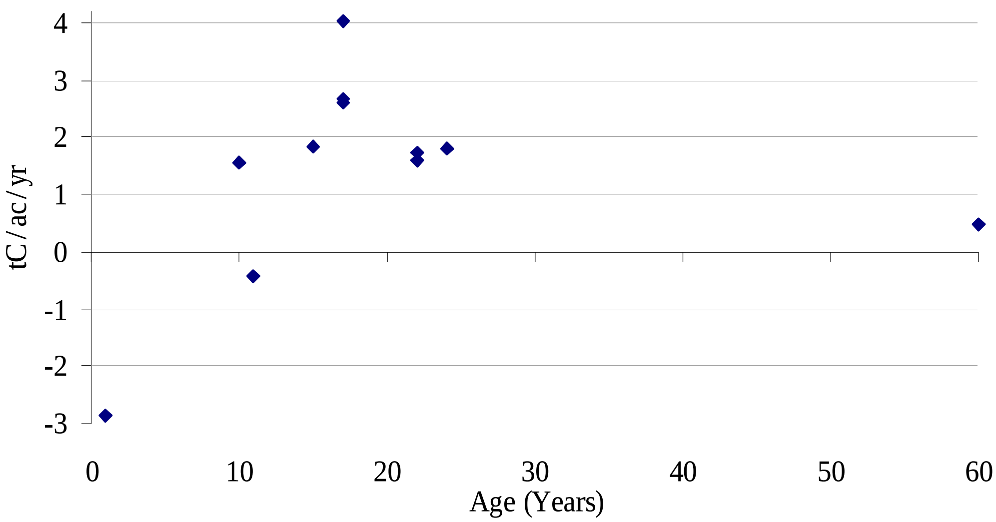


Figure 3.2: Net sequestration values for conifers of different ages from studies in the southeastern U.S., adjusted to GSL in study area (based on Table 3.2)

A 3rd-order polynomial provides the best fit to the GSL-adjusted NEE observations after the 60-yr data point is excluded (Figure 3.3). Inclusion of the 60-yr observation would lead the curve to become negative for years 35 through 55, which is unrealistic. We use this function to estimate carbon NEE for conifers aged 0-23 years.

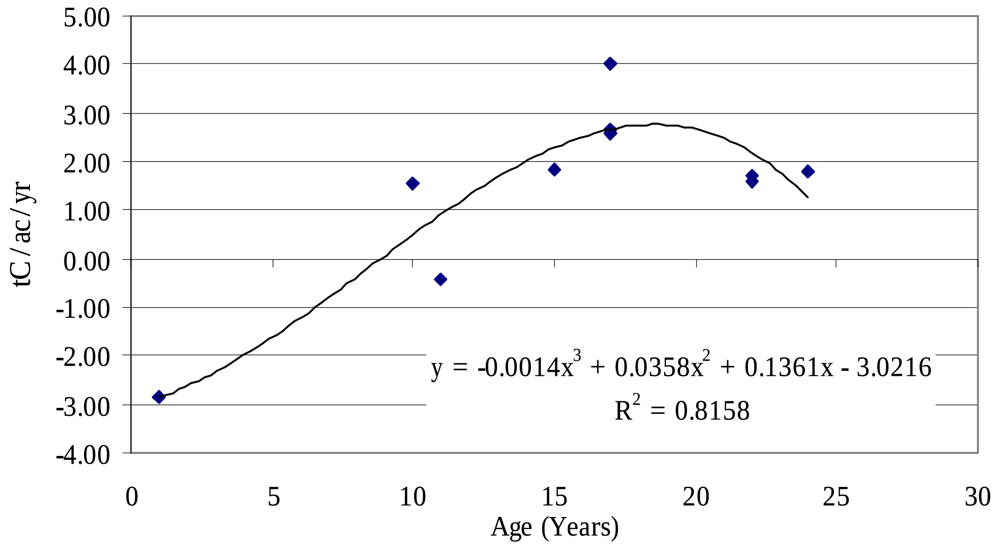


Figure 3.3: Function with the best fit for the GSL-adjusted NEE values

To estimate NEE for conifers aged 24 years and older, we assume that NEE decreases exponentially after age 23 and fit an exponential decay function to the NEE observations for 24 and 60-yr old conifers (Figure 3.4). This assumption is supported by the parabolic curve that describes total aboveground biomass in loblolly pine plantations (Lacatell et al., 2007) and the fact that total aboveground biomass and coarse root biomass of loblolly pine plantations show a constant relationship (Johnsen et al., 2004).

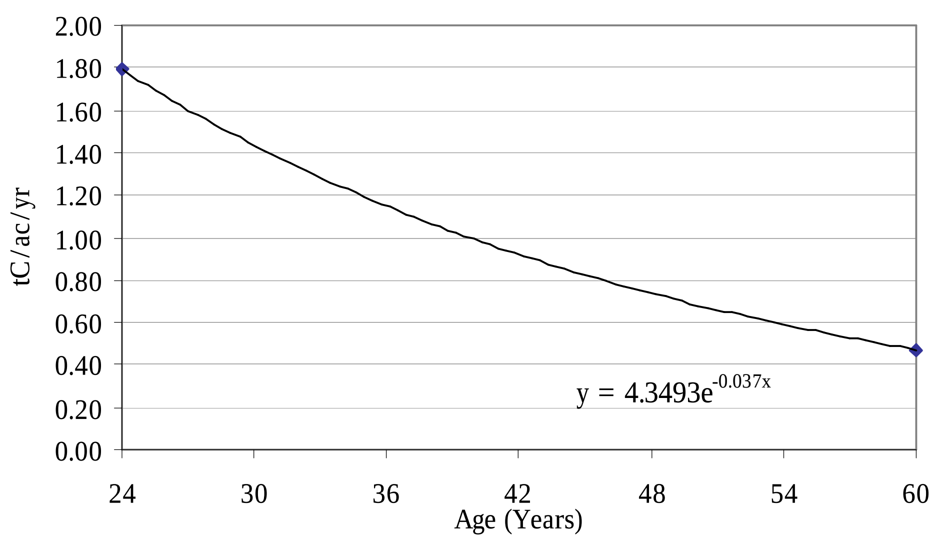


Figure 3.4: Estimated NEE function for conifer trees aged 24-60 years

Because of the assumptions needed to generate aggregate our NEE estimates for conifers, we develop a low and a high conifer NEE estimate. Both estimates are based on the assumption that natural conifer forests account for half of all conifer timberlands, which in 2000 was their share in the state's Northern Coastal Plain (Conner, 2003).[8] Both estimates also assume that the stand age class distribution of conifers in the study area mirrors the distribution in North Carolina's northern coastal plain region as a whole (Figure 3.1).[9] They further assume that within each age class, all trees are distributed evenly over all age classes. Finally, in both estimates, we truncate the age distribution of natural pine forests at 100 years (81+ years age class).

Our high estimate assumes that carbon NEE of natural forests is the same as that of plantations. This is likely to overestimate NEE by natural forests. For example, Johnsen et al. (2001) use plantation estimates as an upper bound on sequestration by all forests.

Our low NEE estimate differs from our high estimate in that it assumes that carbon NEE of natural pine forests is 32 percent lower than that of pine plantations. This assumption is based on Leggett and Kelting (2006), who found that in the East Gulf Coastal Plain in Alabama and Mississippi (mean annual temperature 19°C), fertilization of loblolly pine stands at planting increased total ecosystem carbon content by 32 percent on both sandy and clay soils. Johnsen et al. (2001) review six fertilization experiments that resulted in far larger biomass increases than those reported in Leggett and Kelting (2006). However, these consist of a range of applications, including annual fertilization and weed treatments. Such intensive management is unlikely to represent the typical approach applied on most plantations. Thus, we do not use the treatment impacts reported in that study in our estimates.

By reducing NEE for natural pine forests, our low estimate implicitly assumes that the NEE values reported for plantations (Table 3.2) all are for plantations that are fertilized. Although industrial southern pine forests are increasingly fertilized (Johnsen et al., 2001), it is unlikely that fertilization occurs on all plantations. Thus, assuming that the NEE of natural pine forests is 32 percent lower than that of plantations may result in an underestimation of NEE for natural forests.

Using the methodology described above, we estimate that the conifer lands in our study area sequester a total of between 126 thousand and 145 thousand tons of carbon per year (Table 3.3), or between 464 thousand and 533 thousand tons of CO_2e.

Table 3.3: High and low estimates of aggregate annual carbon Net Ecosystem Exchange of conifer forests

		NEE *tC/yr*	
	Conifer plantations	*Natural conifer forests*	*Total*
High Estimate	86,781	58,631	145,412
Low Estimate	86,781	39,869	126,650

Notes: Based on assumptions as described in text and total conifer acreage in study area as shown in Table 3.1.

[8] At the state level, plantations account for two-fifths of all conifer lands (Brown et al., 2006).

[9] It appears unlikely that the age class composition in the study area is very different from that in the state's northern coastal plain as a whole; after all., the age class composition at the state-level is very similar to that of the northern coastal plain (Brown et al., 2006). Thus, conifer plantations in the area would need to be very atypical in their age class distribution in order for this assumption to be invalid.

The values presented in Table 3.3 represent *estimates* of the average quantity of carbon sequestered by conifers in the study area per year. These estimates are characterized by a degree of uncertainty due to uncertainty in the original data (Table 3.2) and the methods used to adjust these data to the study area. First, interannual variability in growing conditions and spatial variability of NEE even within uniform plantations (Oren et al., 2006) can introduce uncertainty into annual NEE estimates for a given location. However, since most of the values reported in Table 3.2 are multi-year averages, this source of uncertainty should be less of a concern for our analysis. Uncertainty in our NEE estimates also stems from the fact that some of the growing conditions, other than growing season length, for which we make appropriate adjustments, may differ from those found at the sites studied in the literature. However, the fact that the study area and surrounding lands are home to many conifer plantations seems to indicate that the area is well-suited to pine production. Thus, it appears unlikely that the productivity of the area with respect to growing pines is lower than that of the sites reported in the literature (Table 3.2). Overall, our estimates should represent reasonable approximations of the true carbon net sequestration provided by the conifer timber lands in the study area.

Deciduous Forest

Deciduous Forests include oak forest and Coastal Plain mesic forest, both of which are found in the northwestern and southwestern portions of our study area. They also include coastal floodplain forests, which comprise levee forests, cypress gum swamps, bottomland hardwoods, and alluvial floodplains with small poorly defined fluvial features, all of which are present throughout the study area (North Carolina Wildlife Resources Commission, 2005).

There are fewer published studies on the net carbon sequestration by southeastern hardwoods compared to conifers (Table 3.2). In addition, all of these are for stands aged between 60 and 120 years. Thus, unlike in the case of conifers, we cannot use GSL-adjusted NEE estimates from the available studies to estimate an NEE function that covers the lower end of the range of hardwood age classes found in our study area (Figure 3.5).

However, an analysis of red oak stands at Harvard Forest in Massachusetts and of six populations of red oak stands (*Quercus rubra*) in western New York, northern New Jersey and Massachusetts found that the biomass increments accumulated in these stands, with median ages ranging from 72 to 210 years, have been relatively flat over time (Pederson et al., 2003).[10]

Several studies have confirmed that even old, late-succession hardwood forests continue to act as large net carbon sinks (Knohl et al., 2003; Luyssaert, 2008). We assume that the generally linear biomass increment reported in Pederson et al. (2003) also characterizes the hardwoods stands in our study area, and thus apply the GSL-adjusted NEE values reported for southeastern deciduous forests (Table 3.2).

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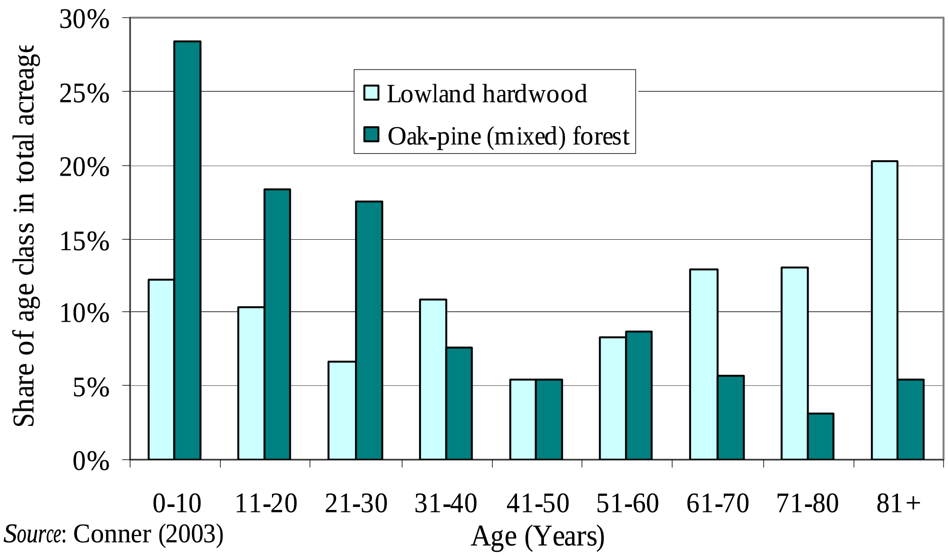
[10] As Pederson et al. (2003) point out, the whole-sample biomass of the two Harvard Forest red oak plots with median ages 72 and 103 years, respectively, shows a linear increase except in the fits years of stand establishment, and a comparison of the youngest and oldest tress shown nearly identical growth rates.



Source: Conner (2003)

Figure 3.5: Age class distribution of lowland hardwood and mixed forests in North Carolina's Northern Coastal Plain, 2000

Because of the range of values reported in the literature, we construct low and high estimates of the aggregate annual NEE of deciduous forests. The low estimate is based on the average of the low and high biometric NEE estimates reported in Hanson et al. (2004); the high estimate is based on the value reported by Greco and Baldocchi (1996) (Table 3.2). The literature estimates are for natural forests; all deciduous forests in North Carolina's Northern Coastal Plain are natural as well (Conner, 2003). Based on these assumptions, we estimate the annual net carbon sequestration by deciduous forests in the study area at approximately 20 thousand to 48 thousand tons of carbon, or 73 thousand to 175 thousand tons of CO_2e (Table 2.4).

Table 3.4: Estimated total annual net carbon sequestration by deciduous forests in study area

	NEE *tC/yr*
High Estimate	19,772
Low Estimate	47,596

Notes: Based on assumptions as described in text and total deciduous acreage in study area as shown in Table 2.1.

Mixed Forest

Mixed forests are found primarily along the northern and southern fringes and the western portion of the study area, the mixed forest category includes oak-pine forests and mesic mixed hardwood forest (North Carolina Wildlife Resources Commission, 2005). Most oak-pine forests in North Carolina's Northern Coastal Plain are natural (Conner, 2003).

Our net sequestration estimates for the study area mixed forests are based on the assumption that oak and pine each account for half of the total acreage overall and within each stand age class (Figure 3.5). To estimate the net carbon uptake by the pine portion of these forests, we apply the carbon NEE function estimated above and the assumptions described in the estimation of the conifer carbon balance.

To incorporate the range of estimates reported in the literature, we develop low and high NEE estimates. Our low estimate is based on the assumption that the NEE of conifers is 32 percent lower than estimated by our conifer NEE function (for reasons discussed in the conifer NEE section above), and applies the average of Hanson et al.'s (2004) low and high biometric NEE estimates for oaks (Table 3.2.) for the oak portion of the mixed forest acreage. Our high estimate does not reduce conifer NEE estimates by 32 percent and applies Greco and Baldocchi's (1996) hardwood NEE estimates to the oak portion of the mixed forests. The methodologies thus are the same as those used to develop NEE estimates for the conifer and deciduous forest cover types, respectively.

Based on these assumptions, we estimate that mixed forests in the study area have an aggregate net carbon balance of between nine thousand and 19 thousand tons per year equivalent to 33 thousand and 70 thousand tons of CO_2, respectively (Table 3.5).

Table 3.5: Estimated total annual net carbon sequestration by mixed forests in study area

	NEE *tC/yr*
High Estimate	9,038
Low Estimate	19,151

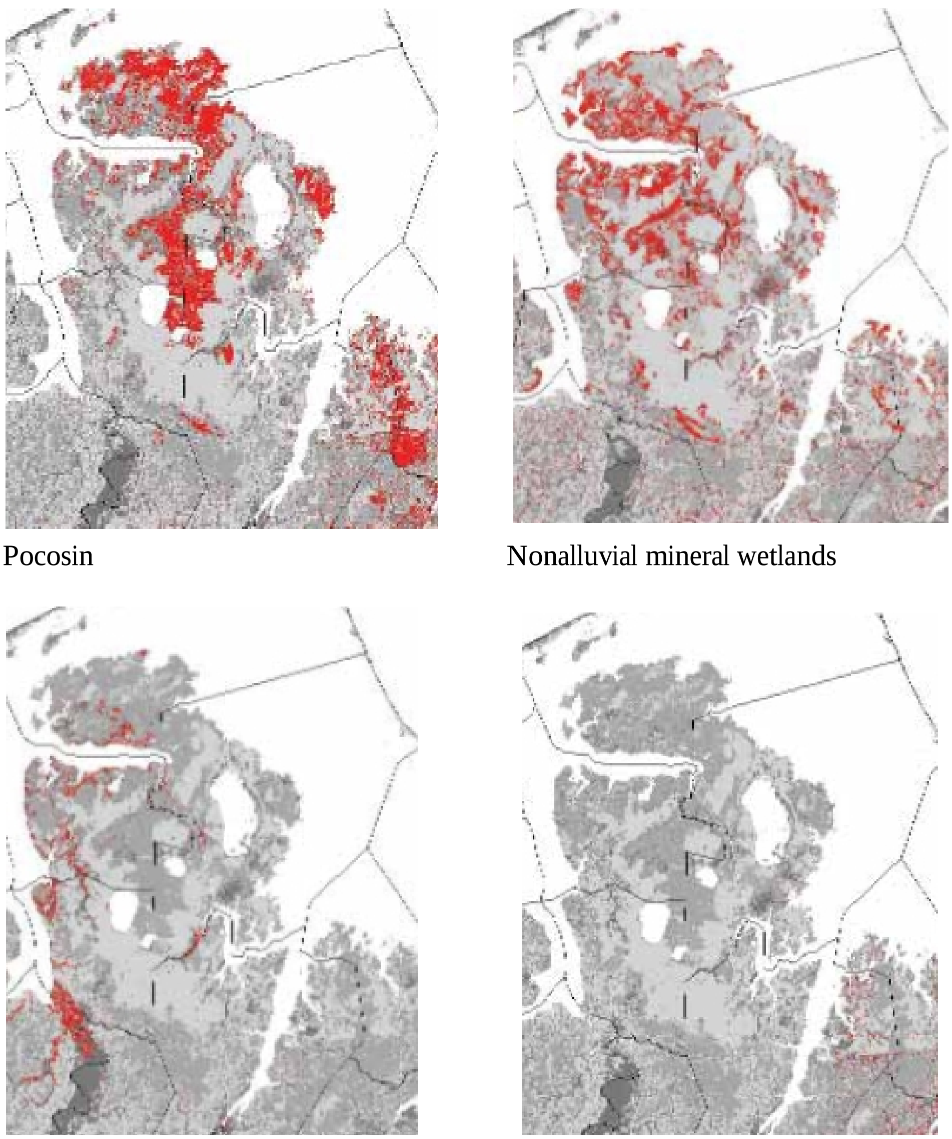
Notes: Based on assumptions as described in text and total mixed forest acreage in study area as shown in Table 3.1.

Woody Wetlands

Woody wetlands are the dominant wetland type found in the study area. Woody wetland types include Pocosin, non-alluvial mineral wetlands, tidal swamp forest and wetlands, and wet pine savannah. Pocosins are peatland communities of the Coastal Plain and include low pocosin, high pocosin, pond pine woodlands, peatland Atlantic white cedar forest, bay forest, streamhead pocosin, and streamhead Atlantic white cedar forest (North Carolina Wildlife Resources Commission, 2005).

In much of the Coastal Plain, the condition of pocosin habitats is poor due to fire suppression, changes in hydrology, intensive silviculture and conversion of forest types. However, extensive examples of low and especially high pocosins still exist on public lands including much of the Albemarle-Pamlico peninsula and many other places. Extensive examples of pond pine woodlands exist in the Green Swamp, at Alligator River National Wildlife Refuge, Pocosin Lakes National Wildlife Refuges and in Dare County at the Dare Bombing Range. Atlantic White cedar dominates in some remaining pocosins where fire is infrequent, but its overall abundance has been greatly reduced by lack of fire, logging, and drainage. Atlantic white cedar-dominated communities still exist at Alligator River and Pocosin Lakes National Wildlife Refuges, and in the Great Dismal Swamp (North Carolina Wildlife Resources Commission, 2005).

Nonalluvial mineral wetlands include a variety of vegetation depending on water saturation and soil type. After Pocosin, they are by far the most ubiquitous wetland type in the study area (Figure 3.6). In the wettest areas, bald cypress, swamp black gum, and red maple dominate these systems. Where nonalluvial mineral wetlands transition to peatland, loblolly pine, pond pine, and Atlantic white cedar may also be present. In less saturated nonalluvial wetlands, trees characteristic of bottomland hardwood systems - cherrybark oak, laurel oak, swamp chestnut oak, tulip poplar, sweetgum, American elm and red maple - dominate (North Carolina Wildlife Resources Commission, 2005).



Pocosin

Nonalluvial mineral wetlands

Tidal swamp forest and wetlands

Wet pine savanna

Figure 3.6: Distribution of the four major wetland types found in the study area (North Carolina Wildlife Resources Commission, 2005)

Tidal swamp forest and wetlands make up a much smaller share of wetlands in the study area than pocosins are nonalluvial mineral wetlands. They occur along rivers or sounds in areas where flooding is influenced by tides. Vegetation may range from Cypress-Gum swamps, characterized by swamp black gum, water tupelo, and bald cypress, to freshwater marshes containing giant cordgrass, sawgrass, cattails, American threesquare, black needle rush, spike-sedges, southern wildrice, arrowhead and marsh fern. Regularly flooded herbaceous sites are reported to have high productivity, equivalent to salt marshes (North Carolina Wildlife Resources Commission, 2005).

Wet pine savannah accounts for an even smaller share of total wetland area in the five counties. It includes several types of mineral wetlands and in the study area is limited to small sections in southern Beaufort County. These communities are characterized by longleaf pine, sometimes mixed with loblolly and pond pine, with herb- or shrub-dominated understory.

Wetlands can be important sinks for atmospheric CO_2, sequestering amounts per unit area similar to those taken up by forests (Table 3.2). However, many wetlands also emit large quantities of methane (CH_4), a greenhouse gas with a global warming potential (GWP) much higher than that of CO_2. A recent comprehensive review concluded that, with the exception of estuarine wetlands, methane emission from wetlands may largely offset benefits from carbon sequestration in wetland soils and plants in terms of their climate impacts (Bridgham et al., 2006). Thus, it is important to include CH_4 in an analysis of the net carbon balance of wetlands (Limpens et al., 2008; Bridgham et al., 2006).

Of the original wetland studies reported in Table 3.2, only Li et al. (2004) include CH_4-based carbon in their net carbon flux analysis. (The net carbon flux figure reported for that study in Table 3.2 does not include CH_4-carbon in order to make the wetland emission estimates reported in the table consistent.) To incorporate the CH_4-based carbon emissions of our study area wetlands in our net carbon balance analysis, we identified methane flux studies of similar wetlands in the literature. Some of these studies were carried out in the North Caroline Coastal Plain. However, the majority is from other areas. Applying these estimates to our study area wetlands is likely to introduce errors into our carbon flux estimates because CH_4 emissions vary considerably among wetland types and locations. Table 3.6 shows CH_4 emission estimates reported in the literature for the wetland types found in our study area.

In order to account for the higher relative global warming impact of CH_4 per unit of carbon compared to CO_2, we convert the CH_4 emissions reported in the studies listed in Table 3.6 to their CO_2 equivalents (CO_2e) based on the standard 100-year time frame GPW of methane of 25.[11] The CH_4 fluxes reported in the table are net fluxes that indicate the balance of CH_4 production and oxidation at the studied wetland sites. Table 3.6 also shows the carbon emissions contained in these CO_2-equivalent emissions (indicated in the table in the C** column), which is the unit of measure of the net CO_2-carbon balance of wetlands reported in Table 3.2. Because CH_4 emissions are a function of temperature and gross plant productivity, among other factors, we adjust the methane emissions reported in these studies for the GSL differential between the source study location and our study area, using the methodology described in the coniferous forest section above and used to generate GSL-adjusted net CO_2-based carbon fluxes reported in the last column of Table 3.2.

[11] Forster et al. (2007).

Table 3.6: Methane and equivalent CO_2 net emission balances reported for selected wetlands

Wetland type	*Location*	*CH_4 t/ac/yr*	*CO_2e * t/ac/yr*	*C ** t/ac/yr*	*Source*
Swamp forest	FL	0.102	2.54	0.69	Bartlett et al. (1989)
Cypress/tupelo swamp forest	LA	0.215	5.37	1.47	Alford et al. (1997)
Open water swamp	FL	0.707	17.68	4.82	Schipper and Reddy (1994)
Wetland forest	FL	0.087	2.18	0.59	Harriss et al. (1988)
Maple/gum forested swamp	VA	0.002	0.05	0.01	Harriss et al. (1982)
Cypress swamp-flowing water	FL	0.099	2.47	0.67	Harriss and Sebacher (1981)
Cypress swamp - deep water	GA	0.136	3.40	0.93	Harriss and Sebacher (1981)
Cypress swamp-floodplain	SC	0.015	0.36	0.10	Harriss and Sebacher (1981)
Cypress swamp-floodplain - saltwater	SC	0.002	0.05	0.01	Bartlett et al. (1985)
Emergent tidal marsh	VA	0.389	9.74	2.66	Neubauer et al. (2000)
Emergent macrophytes (Peltandra)	VA	0.228	5.71	1.56	Wilson et al. (1989)
Emergent macrophytes (Smartweed)	VA	0.122	3.06	0.83	Wilson et al. (1989)
Emergent macrophytes [1]	OH	0.005	0.13	0.04	Altor and Mitsch (2008)
Emergent macrophytes [2]	OH	0.044	1.09	0.30	Altor and Mitsch (2008)
Tidal freshwater wetland forest	NC	0.004	0.10	0.03	Megonigal and Schlesinger (2002)
Tidal freshwater wetland forest	NC	0.009	0.23	0.06	Megonigal and Schlesinger (2002)
Pine-spruce wetland	FL	0.318	7.94	2.17	Li et al. (2004)
Forested peatland	NY	0.001	0.02	0.01	Coles and Yavitt (2004)
Short pocosin	NC	0.004	0.09	0.03	Bridgham and Richardson (1992)
Tall pocosin	NC	0.002	0.05	0.01	Bridgham and Richardson (1992)
Gum swamp	NC	0.009	0.22	0.06	Bridgham and Richardson (1992)

Notes: All studies are for freshwater systems except where indicated otherwise. Except in the case of Altor and Mitsch (2008), Li et al. (2004), Bridgham and Richardson (1992) and Coles and Yavitt (2004), CH_4 emissions are the used annual fluxes reported in Appendix 1 in Bridgham et al. (2006). * Based on methane GWP of 25 (Forster et al., 2007). ** Equivalent units of carbon contained in the emitted CH_4, expressed at GWP of CO_2. The CO_2:C molecular weight ratio is 3.667. [1] Minimum of four sampled emergent wetlands. [2] Maximum of four sampled emergent wetlands.

To develop estimates of the overall CO_2 and CH_4 net carbon balance of the wetlands in the study area, we construct low and high estimates for CO_2 and CH_4 emissions for each of the major wetland types found in the study area. Where more than one estimate is available for the net CO_2 or CH_4 balance of a particular wetland type, we use low and high estimates for each gas to construct low and high estimates of the combined carbon dioxide and methane net balance for specific wetland types. The low combined CO_2 and CH_4 net carbon balance is derived by subtracting the high estimate of net CH_4 emissions, converted to their CO_2-C equivalent using methane's GWP of 25, from the low estimate of net CO_2 flux. Conversely, the high combined CO_2 and CH_4 net carbon balance estimate is derived by subtracting the low estimate of net CH_4 emissions, converted to their CO_2-C equivalent, from the high estimate of net CO_2 flux. The low and high CO_2 and CH_4 values used in the estimates are shown in Table 3.7.

Table 3.7: Values used in constructing low and high combined net carbon balances for the wetland types found in the study area

	Net CO_2-C balance; GSL-adjusted tC/ac/yr		*Net CH_4-C balance; GWP-adjusted tC/ac/yr*		*Net CO_2-C - net CH_4-C; GSL and GWP-adj. tC/ac/yr*	
	Low	High	Low	High	Low *	High*
Pocosin	0.064 [a]		0.013 [b]	0.025 [c]	0.038	0.050
Wet pine savanna	1.131 [d]		1.421 [d]		-0.290	
Nonalluvial mineral wetlands	0.069 [e]	0.516 [f]	0.059 [g]	0.360 [h]	-0.291	0.458
Tidal swamp and forest wetlands	0.413 [j]		0.028 [k]	0.063 [k]	0.349	0.385
Emergent herbaceous	0.739 [l]	8.512 [m]	0.053 [n]	1.719 [o]	-0.980	8.459

Notes: Positive values indicate a net carbon sink; negative, a net carbon source. CO_2-C - carbon contained in carbon dioxide; CH_4-C - carbon contained in methane; GSL - growing season length; GWP - Global Warming Potential. * Low total CO_2 -CH_4 net balance: low CO_2 net sequestration minus high CH_4 net emissions. High total CO_2 -CH_4 net balance: high CO_2 net sequestration minus low CH_4 net emissions. [a] FWS (2009). [b] Tall pocosin, Bridgham and Richardson (1992). [c] Short pocosin, Bridgham and Richardson (1992). [d] Li et al. (2004). [e] Average of North American freshwater mineral wetlands, from Bridgham et al. (2006). [f] Average of all wetlands in Table 3.2, due to wide range of vegetation types falling into this category pin the study area (North Carolina Wildlife Resources Commission, 2005). [g] Gum swamp (Bridgham & Richardson 1992). [h] Geometric mean of nonalluvial values in Table 3.6, excluding peatlands. [j] Craft et al. (2006). [k] Megonigal and Schlesinger (2002). [l] Heinsch et al. (2004). [m] Net uptake of most productive of four created emergent wetlands examined in Altor and Mitsch (2008). [n] Lowest-net-emitter of four created emergent wetlands examined in Altor and Mitsch (2008). [o] Higher-emitting of the two wetland areas examined in Wilson et al. (1989).

Our overall net carbon balance for woody wetlands is derived by multiplying the low and high per-acre net balance estimates of the different woody wetland types (pocosin, wet pine savannah, nonalluvial mineral wetlands and tidal swamp and forest wetlands) with the total acreage of each of these wetland types, based on their respective shares (Figure 3.5) in total woody wetland acreage.

Emergent Herbaceous Wetlands

Emergent herbaceous wetlands in our study area occur in the form of freshwater marshes. Estimates of their CO_2 and CH_4 net fluxes are presented in Tables 3.2, 3.6, and 3.7. Combining the low CO_2 net flux estimates with the high CH_4 net flux estimates in the low scenario and the high CO_2 and low CH_4 estimates in the high scenario yields the widest possible range of net carbon balances. Therefore, it is probable that the actual net combined CO_2 and CH_4 carbon balance falls somewhere between these two extremes.

Overall wetland carbon balance

The net carbon (CO_2 and CH_4) balance estimates for the different wetland types are shown in Table 3.8. The estimates show that overall, wetlands in the area are expected to act as a carbon sink. However, based on our analysis, this is not true for wet pine savannas. Furthermore, the net carbon balance of nonalluvial mineral and emergent herbaceous wetlands may be positive, indicating a carbon net source, or negative, indicating a net carbon sink. Nevertheless, the mean of the low and high estimates for the two wetland types is positive, suggesting that both are expected to act as net absorbers of atmospheric carbon.

It bears emphasizing that our estimate for wet pine savanna is based on a single study of a pine-spruce wetland site in northern Florida (Li et al., 2004). The same study reported a net carbon uptake by a similar wetland type in Minnesota. In addition, the ranges of carbon estimated balances for nonalluvial mineral and emergent herbaceous wetlands are very large, as a result of the large differences in estimates reported in the literature.

Table 3.8: Estimates of the combined annual net CO_2 and CH_4 carbon balance of wetlands in study area

	Low estimate	*High estimate*
	tC/yr, at GWP=1	
Woody wetlands		
Pocosin	14,127	18,563
Wet pine savanna	-10,699	
Nonalluvial mineral wetlands	-85,962	135,130
Tidal swamp and forest wetlands	12,893	14,218
Emergent herbaceous	-135,737	1,171,901

Notes: Positive values indicate a net carbon sink; negative, a net carbon source. Estimates show CH_4-carbon converted to its CO_2-equivalent based on $GWP_{100\ year}$. All values are in weight of carbon (C), which is 1/3.667 the weight of the corresponding quantity of CO_2. Acreage of woody wetland types based on total woody wetland acreage from Table 3.2 and relative extent of the four woody wetland types shown in Figure 3.5.

Finally, to the extent that some of the remaining wetlands in the study area are impacted by human activity, their carbon balance may be changed. For example, the lowering of the water level on peatlands as a result of the draining of surrounding agricultural lands or the peatlands themselves will impact the carbon balance of peatlands, producing large and long-lasting carbon releases (Bridgham et al., 2006; Limpens et al., 2008). This impact of human activity on the size and sign of carbon fluxes is also true for other wetland types (Li et al., 2004). However, our estimates are based on reported carbon balances of more or less intact wetland systems. Thus, to the extent that the wetland acreage in the study area, as identified in the NLCD data and the North Carolina Wildlife Action Plan (North Carolina Wildlife Resources Commission, 2005), is impacted by human interventions, our wetland carbon balance estimates may be inaccurate. This concern is especially relevant for peatlands that border on private lands that have been converted for agricultural or forestry use. Some of the large peatland complex found on National Wildlife Refuge (NWR) lands in the study area is being impacted by historic changes in the hydrology on neighboring private lands, even though some of those lands are no longer used for agriculture or forestry (FWS, 2009). To date, hydrological restoration has been completed on almost 11,000 acres of severely drained areas of the Pocosin Lakes NWR and will be completed on the remaining 5,000 severely drained acres, with an additional 10,200 acres of less severely drained refuge lands targeted for future restoration (FWS, 2009). The approximately 35,000 acres of degraded pocosin wetlands with restoration potential on Alligator River NWR are targeted for restoration as well (FWS, 2009).

The ongoing efforts to restore the hydrology of disturbed peatlands and surrounding converted private lands will stop the release of carbon from these lands that currently occurs through soil decomposition and oxidation and the release of soil organic matter into waterways and will restart the process of net carbon accumulation on these lands (FWS, 2009).

Because of the much higher uncertainties associated with the net wetland carbon balance compared to the other vegetation types analyzed in this study, the wetland balance estimates should be considered less reliable. The development of reliable estimates will require more detailed site-specific analyses.

Shrubland / Scrubland

Shrublands are defined as being dominated by shrubs and low woody plants generally below three meters in height, with roughly 50 percent or more of the area covered in shrubs or low woody vegetation (North Carolina Center for Geographic Information and Analysis, 1994). Included in this category are areas of immature trees that may be in transition to forest as well as cultivated areas with shrub-like crops, including berries and vine crops. Shrubland habitat is often found at the transition between agricultural fields and nearby woodlands, created by disturbances like clear cutting, disking or burning, and exists throughout much of the study area, with the general exception of the refuges (North Carolina Wildlife Resources Commission, 2005).

The only sequestration estimate for shrublands in the southeast is from Powell et al. (2006), who studied the net carbon flux of a scrub-oak system in eastern central Florida during 2000-2006 (Table 3.2). We adjust their reported net uptake estimate for growing season length differences between their and our study areas using the methodology described in the coniferous forest section. Using the adjusted estimate of net carbon uptake per acre of 0.66 tons per year, the approximately 60 thousand acres of scrub-shrublands in our study area are estimated to have a total net uptake of around 40,000 tons of carbon per year.

Grasslands

Grasslands form part of the early successional habitat in the Coastal Plain Ecoregion and are found throughout much of the study area, though less on the Albemarle Peninsula (North Carolina Wildlife Resources Commission, 2005). We were able to locate only two studies of grassland net carbon sequestration for the southeastern U.S. in literature (Table 3.2). Novick et al. (2004) found that grasslands in Orange County, North Carolina were a net source of carbon, but argued that this was attributable to a drought in the measurement year. Modeling the carbon balance under normal (average) conditions, they found that the grassland was a weak carbon sink. Stoy et al. (2006) used two methods to estimate grassland NEE over a five-year period. One of the methods (short-term exponential fits [STE]) yielded "anomalous" results and estimated that the grassland acted as a carbon source, while the other (non-rectangular hyperbolic [NRH]) suggested that the grassland acted as a carbon sink if the carbon stored in the biomass removed by annual mowing was included in the carbon balance.

We use the NEE values reported in Novick et al. (2004) to construct a low estimate of the net carbon uptake by grasslands in our study area. To account for the impact of drought on grassland NEE, we weight Novick et al.'s (2004) drought year and normal year NEE estimates by the relative frequency of drought and non-drought years, respectively, in the study area to construct and average NEE estimate for grasslands in the area.

From June 1999 to May 2009, portions or all of the study area experienced moderate, severe or extreme drought for 140 of the 518 weeks (Table 3.9), with drought observations (in either part or all of the study area) according for 24 percent of all observations. The drought frequency-weighted average NEE of Novick et al.'s (2004) reported values (including biomass removed during annual mowing) is 0.105 tC/ac/yr.

Table 3.9: Drought conditions in the study area, 1999-2009

	Normal		*D0-abnormally dry*		*D1 Drought-moderate*		*D2 Drought-Severe*		*D3 Drought-Extreme*	
# of	All	Parts	All	Parts	All	Parts	All	Parts	All	Parts
Weeks	319	20	64	38	48	28	16	29	1	18
	Normal/abnormally dry				*Drought (moderate, severe, extreme)*					
# of			All	Parts			All	Parts		
Weeks			383	58			65	75		

Source: Drought Monitor Archives maps for North Carolina (National Drought Mitigation Center, 2009).

Our high estimate of average grassland net carbon sequestration in the study area is based on the average of Stoy et al.'s (2006) NRH and STE-based NEE values (0.543 tC/ac/yr); we include the anomalous STE-based values in order to be conservative. Since the study's five-year measurement period included drought and non-drought years, their reported values already reflect the impacts of droughts.

Based on these NEE estimates, we estimate that the total net carbon uptake by grasslands in the study area is between 8,200 and 42,100 tons of carbon per year.

Estimated total carbon net uptake by study area natural lands

The approximately 1.2 million acres of undeveloped and nonagricultural lands in the five-county study area represent a sizeable carbon sink (Table 3.10). Forests and scrublands in the area together net sequester around 200,000 to 250,000 tons of carbon annually. For grasslands and especially for wetlands, the spread between low and high estimates is far larger. While we estimate that grasslands take up between around 8,000 and 40,000 tons of carbon per year on a net basis, our analysis suggests that wetlands as a whole may be either a net carbon source of a scale comparable to the forest sink, or a net sink about five times as large as the forest sink. The disparity between the low and high wetland sequestration estimates is due primarily to the range of and the sign on the sequestration estimates for nonalluvial mineral wetlands and emergent herbaceous wetlands.

Furthermore, it bears emphasizing that the wetland carbon balance estimates account for the net release of methane from wetlands, weighted at the GWP_{100yr} of methane of 25.

Table 3.10: Estimated annual net carbon uptake by study area vegetation types, not accounting for timber harvests

	Low est.	*High est.*	*Mean est.*
		tC/yr, at GWP=1	
FORESTS AND SHRUBLANDS			
Coniferous forests	39,869	58,631	49,250
Conifer plantations	86,781	86,781	86,781
Deciduous forests	19,772	47,596	33,684
Mixed forests	9,038	19,151	14,094
Shrubland/scrubland	39,613	39,613	39,613
TOTAL FORESTS AND SHRUBLANDS	195,073	251,772	223,422
WETLANDS			
Pocosin	14,127	18,563	16,345
Wet pine savannah	-10,699	-10,699	-10,699
Nonalluvial mineral wetlands	-85,962	135,130	24,584
Tidal swamp and forest wetlands	12,893	14,218	13,556
Emergent herbaceous	-135,737	1,171,901	518,082
TOTAL WETLANDS	-205,378	1,329,114	561,868
GRASSLANDS	8,159	42,107	25,133
TOTAL	-2,147	1,622,993	810,423
TOTAL w/out wetlands	203,231	293,879	248,555

Excluding wetlands from the analysis on the grounds of the uncertainties about the latter's net carbon balance, the remaining undeveloped nonagricultural lands in the study area (forests, shrublands and grasslands) together take up between around 200,000 and 290,000 tons of carbon per year on a net basis, or between 750,000 and 1,080,000 tons of CO_2e.

Adding the net carbon balance of wetlands to that of forests, shrublands and grasslands dramatically changes the picture and introduces major uncertainty, making the nonagricultural vegetation an insignificant source of or a very large sink for carbon of around 1.6 million tons per year (Table 3.10), or close to 6 million tons of CO_2e, due to the uncertainty associated with the net carbon balance of some wetland types.

These numbers do not account for the carbon that is removed from the system in the process of timber harvests. The inclusion of harvests in the analysis dramatically changes the picture.

Accounting for timber harvests

In order to determine the net quantity of carbon actually stored in study area vegetation, it is necessary to account for carbon removed from the system by timber harvests since the latter account for a substantial portion of total carbon stocks in the ecosystem.

It is often argued that wood products carbon be included in offset schemes (Ruddell et al., 2007). Of the total quantity of carbon contained in the cut trees, in the US on average only 18 percent end up in long-lived wood products (Ingerson, 2009).[12] The remainder is accounted for by losses at the harvesting, and primary and secondary processing and construction phases (Ingerson, 2009). The carbon lost at each of these stages is released as carbon dioxide or methane from during the processes of decay and decomposition. Of the 18 percent of live tree carbon that ends up in long-lived wood products, each year about 1.5 percent on average is lost when durable wood products are disposed (Smith et al., 2006; USDOE, 2007). After 100 years (the time frame commonly applied in carbon accounting for climate purpose), on average only one percent of the carbon contained in the standing tree remains bound up in long-lived wood products (ibid.). If the carbon contained in wood products in landfills is included, the portion of carbon dioxide equivalent of the standing tree still contained in long-lived wood products and landfills drops to 14 percent after 100 years on average (Ingerson, 2009).[13] These estimates still overestimate the percentage of live tree carbon that ends up in long-lived wood products as they exclude below-ground carbon releases after harvest.

Connor's (2003, tables 33 and 36) data on softwood harvests in our study area indicates that, on average, during 1990-1999 harvests exceeded net growth by 21%.[14, 15] The picture for hardwoods is similar. Taking into account the fact that of the harvested timber, only 18 percent on average end up in long-lived wood products, the aggregate carbon balance of study area forests outside of conservation areas is clearly negative. Furthermore, this does not account for soil losses and the ongoing additional loss of carbon (in both carbon dioxide and methane form) from the portion stored in long-lived wood products that is discarded each year.

Individual forest lands of course may be net sequesters of carbon. Lands that are not logged or that are logged in such a way that annual net carbon uptake exceeds carbon removal, and lands taken out of timber production and planted as forest or naturally reverting to forest, all increase the carbon stored in the ecosystem.

With forest lands in the aggregate being a source of carbon emissions as a result of timber harvests, the only ecosystem types that are estimated to be net accumulators of carbon in our study area are shrub and scrublands, grasslands and wetlands. The mean estimate of the net uptake provided by these three ecosystems is approximately 625,000 tC/yr (at GWP=1), or 2.3 million tCO_2e.

Social and Market Value of Carbon Sequestration

The value of carbon sequestration has two components. First, sequestration produces social benefits in the form of the avoided damages that the sequestered carbon would have caused had it remained

12 Medium estimate in table 2 in Ingerson (2009).
13 This accounts for the portion of carbon released in the form of methane, which has a global warming potential of 25.
14 We note that our NLCD land cover-based forest acreage estimates vary from the timberland acreage estimates for the five counties as shown in Connor (2003). This difference may be due to slight differences in classifications or to estimation methodologies, with the NLCD dataset based on 30x30m (with a 1 acre minimum) satellite imagery, and Connor's data based on plot samples.

15 Sixty percent of softwood harvests and 100 percent of hardwood harvests in the study area are from natural forests, with the remainder of softwood harvests coming from plantations (Connor, 2003).

in the atmosphere. In addition, as Weitzman (2009) points out, reducing atmospheric concentrations of greenhouse gases (GHGs) may also be thought of as buying insurance against uncertain extreme climate events with catastrophic damages. In contrast, the market value of carbon sequestration is simply the financial value of offsets generated by creditable sequestration activities.

The social and market value of carbon sequestration differ not only in terms of the benefits that are being valued (avoided damages to society at large vs. creditable units of carbon generated by offset providers) but also in the quantities of carbon included in the valuation. For determining the social value of reduced atmospheric concentrations of GHGs resulting from a particular project, what matters are the net reductions in concentrations that a project achieves. By contrast, the market value of a carbon sequestration project depends on the quantities of avoided carbon that are attributed to the project based on the offset calculation guidelines that govern a particular market. Due to differences in carbon accounting guidelines, the creditable quantity of carbon produced by a given project can vary by nearly a full order of magnitude (Galik et al., 2009).

Social value of carbon sequestration

Due to the often substantial uncertainties surrounding the size and timing of specific impacts from rising atmospheric GHG concentrations, estimates of the size of potential damages reported in the literature vary widely. For this reason, any estimates of the benefits from reduced GHG concentrations necessarily are very uncertain. Nevertheless, the Intergovernmental Panel on Climate Change (Parry et al., 2007a) in its Forth Assessment Report reviewed the literature on the social cost of carbon (SCC) emissions and found that the mean estimate of the SCC in pre-2005 studies was $43/tC - around $12/tCO$_2$e - with a large range around this mean (Parry et al., 2007b).

Using this mean estimate of $43/tC, the social value of the carbon sequestered annually by current study area forests (excluding scrub- and shrub lands) would be between $6.7-7.9 million per year – if there were no timber harvests. However, with the amount of carbon lost from the system through harvests, that value becomes negative (even if accounting for the carbon stored in long-lived wood products), indicating that the current management of study area forests is estimated to impose climate externalities that carry social costs. Timber extraction would have to be reduced by around 20 percent to make study area forests climate neutral.[16]

The carbon sequestration services provided by the other non-agricultural lands in the study area (scrub and shrub lands, wetlands, grasslands) carry a combined social value estimated at -$6.8 to $26.9 million per year, with wetlands dominating these estimates.

Market value of carbon sequestration

Not all carbon sequestered by ecosystems in the study area will be creditable on existing or future markets. Whether or not the net carbon uptake on a particular property is creditable depends on several factors. For both regulatory and voluntary carbon markets, creditability depends on whether or not the particular project type (e.g., conifer plantation) is accepted as an offset by the market, and

[16] With current harvests exceeding net growth by around 20 percent (Connor, 2003), such a reduction would still lead to harvest-related carbon losses from the decay of below-ground tree biomass. However, with an estimated 18 percent of tree carbon ending up in long-lived wood products (Ingerson, 2009), the carbon balance of study area forests would be approximately even.

on whether the project characteristics comply with the conditions the regulatory system places on that project type (e.g., plantations of indigenous species only; compliance of project with guidelines for biodiversity protection or other non-carbon goals). In addition, the amount of creditable carbon may or may not be equal to the amount of actual net carbon uptake by the project. For example, creditable carbon may be less than net carbon uptake if a particular carbon market establishes different credit ratios for different types of carbon. For example, offsets in general may be discounted vs. emission reductions, offsets from single-species plantations may be discounted vs. offsets from plantations that mimic the composition of natural forests. In addition, a percentage of earned offsets may not be able to be sold immediately but may have to be deposited in a reserve pool to compensate for catastrophic losses, as in the case of the Chicago Climate Exchange's forest-based offsets. Furthermore, a portion of the calculated carbon may be discounted to account for leakage (the displacement of reduced harvests to other areas) or additionality (Murray et al., 2009).

Currently, the Chicago Climate Exchange (CCX) is the only organized carbon offset market to which landowners in the study area have access. The CCX is a voluntary market on which prices historically have been much lower than on other Kyoto-based markets such as the European Emission Trading Scheme or the Clean Development Mechanism. Prices there also have been more volatile in recent years, reflecting changes in expectations regarding the advance and design of national cap-and-trade legislation. For example, in 2008, prices on the CCX ranged from \$1-\$7.40/tCO_2e, while in 2009, they ranged from \$2.35 to \$0.1/tCO_2e (following the reduction in expectations that climate legislation will pass in the near future); however, as of 2009, most analysts expected carbon prices to increase and reach between \$20 and \$30 per ton of tCO_2e by 2020 (e.g., see New Carbon Finance, 2008; U.S. EPA, 2009) if national-level cap-and-trade carbon legislation is approved. Obviously, recent developments make it appear uncertain whether or not such a system will be implemented in the next several years.

The CCX allows carbon offsets for "sustainably managed forests projects" (Chicago Climate Exchange, 2009a), which it defines as forestry operations whose growth in carbon stocks exceeds harvest volumes. It also allows offsets for long-lived wood products from sustainably managed forests. In addition, afforestation projects and avoided deforestation projects that are contiguous with afforestation projects are eligible for offset generation (CCX, 2009a). Thus, presently only forest-based carbon can be used to generate carbon offsets on the CCX in our study area.[17] However, as already discussed, as an aggregate, the forest lands in our study area do not qualify for offset generation under the CCX's Sustainably Managed Forest Offset Project protocol since harvests exceed net growth. This result may vary for individual lots of course.

Landowners who decided to stop timber extraction from their forests on average could generate an estimated 220 tons of CO_2e net sequestration per acre over 100 years in the case of conifers, starting immediately after the last harvest.[18] At current (March 2010) CCX carbon prices of 10 cents per ton of CO_2e, this would results in negligible earnings from offset sales.[19] However, at prices of \$20 or

[17] The CCX (2009b) also allows offsets for soil carbon sequestration by sustainably managed rangeland projects. However, the counties eligible to participate in that protocol are all located in the Great Plains and the western US.
[18] This number represents NEE and is calculated based on our estimated annual NEE functions presented Figures 2 and 3. It would be higher if soil carbon losses after harvests are not accounted for and instead aboveground carbon accumulation was counted.
[19] This is even more the case given that in calculating offsets, estimated carbon sequestration is discounted by two times the reported statistical error associated with a 90% confidence interval of the baseline inventory data to account for uncertainty in estimation models. On the other hand, if carbon flows are based on annual in-field inventories, no discounting is applied (CCX, 2009a).

\$30 per ton of CO_2e, this situation changes dramatically. Even so, offset revenues may still not be competitive with forestry. In addition, at least currently, the CCX would not accept this kind of project, as only former forest land that has been in non-forest use for at least ten years is eligible for generating afforestation offsets. Avoided deforestation projects could generate higher offset income because, depending on the age of the standing forest, because carbon stocks in mature forests exceed those of regrowing forests.

In addition to the CCX, voluntary "over-the-counter" (OTC) markets handle deal-by-deal transactions of more tailored offsets than those that take place in the CCX OTC markets accept afforestation and reforestation credits as well as credits from many other projects (Hamilton et al., 2009). In 2008, the U.S. was home to the majority of OTC afforestation and reforestation projects worldwide (ibid.). The volume-weighted average OTC prices in 2008 for forest-related projects in the U.S. ranged from \$6.3-7.7 per ton of CO_2e (Table 3.11).

Table 3.11: Volume-weighted average OTC carbon offset prices for forest projects in 2008

Project type	*Price per ton of CO_2e (2008\$)*	*Price per mt C (2008\$)*
Afforestation/Reforestation Plantation	6.4	25
Afforestation/Reforestation Conservation	7.5	27.5
Forest Managemt.	7.7	28
Avoided deforestation	6.3	23

Source: Hamilton et al. (2009)

Note that within these project categories, prices can vary substantially, depending among other things on the particular standard a project uses for certification (e.g., Gold Standard, Social Carbon, Voluntary Carbon Standard [VCS], etc.) (Hamilton et al., 2009).
Thus, currently, private landowners often should be able to generate higher income from OTC projects compared to registering their project with the CCX. Participation in either market involves transaction costs on the part of the landowner, although these can be lowered substantially if the landowner markets their project through an aggregator, such as the Conservation Trust for North Carolina.

It is important to note that even in cases where afforestation and avoided deforestation offset revenue in itself is not competitive with forestry, in combination with earning from other compatible uses such as hunting leases and perhaps water quality credits, it might well add sufficient additional revenue to make land conservation or restoration, including of red wolf habitat, competitive with timber production. Estimating the actual income stream from carbon offsets generated through the conservation of forest land for particular properties requires a property-specific analysis. Land owners can obtain the information necessary for the generation of such estimates from local aggregators.

IV. Residential Property Value Premiums and Open Space Amenities

Our five-county study area contains over 1.2 million acres of terrestrial natural lands that are not in residential or agricultural use. Of these, approximately one million acres are in forest, forested wetlands and shrub- and scrublands, with much smaller shares for pasture and herbaceous wetlands (Table 4.1). Evidence from a large volume of studies suggests that proximity to natural open space increases the values of nearby properties. The open space property value premiums attributable to the natural lands constitute one of the benefits produced by these lands. In this section, we focus on those natural lands located within one mile of residential properties.

The increment in value a property receives due to its proximity to open space is variously referred to as the open space property value premium, the property enhancement value, or the amenity premium. This premium is the result of what Crompton (2001) calls the proximate principle, namely, the general observation that the value of an amenity is at least partially captured in the value of properties in proximity to that amenity. The idea underlying the proximate principle is that a property, like any good, may be thought of as a bundle of attributes (Lancaster, 1966). The price of the good reflects the value consumers assign to that bundle of attributes. In the case of a property, these attributes include the physical characteristics of the property itself and of any structures and includes property size, relative land scarcity, size and quality or age of structures, neighborhood characteristics such as schools, public safety, and environmental amenities associated with scenic views, clean air, or recreation opportunities. If people value open space and the amenities associated with it, then these values to some extent should be reflected in property prices.

The evidence in the published literature for the existence of the property enhancement value of open space is strong. There are over 60 published articles in the economics literature that examine the property enhancement value of open space (McConnell and Walls, 2005). A number of recent literature reviews have been conducted on the topic. Some of these cover various types of open space, including forest lands, parks, coastal and inland wetlands, grasslands, and agricultural lands (e.g. Fausold and Lillieholm, 1999; Banzhaf and Jawahar, 2005; McConnell and Walls, 2005). Other studies are specific to particular types of open space such as parks (Crompton, 2001), wetlands (Brander et al., 2006; Boyer and Polasky, 2004; Heimlich et al., 1998), or agricultural lands (Heimlich and Anderson, 2001). These findings show that in general, there appears to be an inverse relationship between the scarcity of open space and its property enhancement value, suggesting that open space is relatively more valuable where it is in relatively short supply (McConnell and Walls, 2005).

This of course does not mean that property premiums do not exist in rural areas. As Ready and Abdalla (2005) note in response to a reviewer’s comments, it is theoretically plausible that individuals’ willingness to pay (WTP) for open space could be higher in suburban or rural areas, because residents locate there specifically due to their strong preferences for open space. There are a number of studies in rural areas that do show that open space does indeed increase property values considerably (Phillips, 2000; Vrooman, 1978; Brown and Connelly, 1983; Thorsnes, 2002). These studies generally involve public open spaces that often are comparatively large and enjoy a high level of protection from development, including state parks, forest preserves, and wilderness areas. The large open spaces in our North Carolina study area include several National Wildlife Refuges, and many of the unprotected spaces in the area appear not to be under near-term pressure from development. Thus, many of the open spaces in the area share the characteristic of an expected “permanence” with large protected open spaces. Previous research suggests that it is this permanence of an open space rather than the protected status itself that people value (Earnhart, 2001, 2006), we expect that the large open spaces in the North Carolina study area are not intrinsically less attractive to nearby residents than if they were officially protected.

Open space is not a homogenous good, and the particular attributes of a given open space can be expected to influence the size of the associated premiums received by nearby properties. This is confirmed by the large range in open space premiums (measured as a share of the total value of a property) found in the literature. Table 4.1 summarizes the findings reported in the literature on how particular study area characteristics influence open space premiums.

Table 4.1: Variables that influence the property enhancement value of open space

Variable	*Direction of influence*
Scarcity of open space	+
Protected status/permanence	+
Size of open space	+
Distance to open space	- *
Type of open space	+/-
Opportunity costs / value of competing land uses	+
Income	+

Notes: * Exception: In cases of heavily used public open spaces such as some urban parks, adjacency to such areas may lead to a loss in privacy for some properties and to an associated negative open space premium on properties adjacent to the park.

Source: Kroeger et al. (2008)

No study on the open space premiums of property values exists for our study area. However, one study (Bin and Polasky, 2005) examined the impact of wetlands on residential property values in rural Carteret County, North Carolina. The authors found that proximity or size of the nearest wetland and wetland percentage within a quarter mile of a property were all negatively related to property values. These findings contrast markedly with the findings of positive wetland impacts on house prices in urban areas (e.g., Mahan et al., 2000; Doss and Taff, 1996). With wetlands accounting for fully 45 percent of the total land area in their study county, Bin and Polasky (2005) attribute their results to the low relative scarcity of wetlands in their study area. This hypothesis is supported by Lupi et al.'s (1991) findings that wetlands were relatively more valuable in areas where they were relatively scarce.

In situations where no original studies are available on the value of the benefits produced by environmental amenities like open space, benefits transfer is a possible tool for inferring the value people assign to these benefits. Benefits transfer is a technique in which researchers estimate the value of particular benefits for a site of interest by using the results of existing studies of similar sites (Loomis, 2005). The validity of the resulting transfer-based estimate depends on the similarity of the sites and user groups. The context-dependence of open space premiums calls into question the validity of using a particular open space premium reported in the literature as an indicator of the premiums received by properties in a different area. Because no original study exists for the study area or an area that would appear to be similar in terms of its physical characteristics and ownership, application of either point or average value based benefits transfer approaches to estimate the property value premiums would be questionable. This leaves meta-analysis-based benefits transfer as a possible approach. Meta-analysis is a statistical technique that uses regression analysis of the findings of several empirical studies to systematically explore study characteristics as possible explanations for the variation of results observed across primary studies (Brouwer, 2000; U.S. Environmental Protection Agency, 2000). The values of key variables from the policy case then are inserted into the estimated benefit function to develop policy-site-specific value estimates. One such meta-analysis of open space property value premiums is available in the literature (Kroeger et al., 2008).

Kroeger et al. (2008) conducted a meta-analysis of 21 original quantitative studies in the U.S. containing a total of 55 observations of open space impacts of conserved lands on property values.[20] They included only those studies that examined predominantly "natural" open spaces, excluding crop lands and heavily-developed urban recreational areas. Their estimated meta-analysis-based regression function has the following form[21]:

(eq. 1)

$$\begin{aligned} P_{OS} &= -6.5903 + 0.4221 * \%OSChange - 0.0068 * \%OSChangeSquared + 2.7619 * FOR \\ &\quad + 1.677 * PARK - 2.7367 * AG + 3.5067 * PROT + 5.3409 * PRIV \ , \end{aligned}$$

where P_{OS} is the open space property premium in percent, *%OSChange* is the percentage of the area within a given radius of a property that is occupied by the open space in question, *FOR* is an indicator (dummy) variable set at 1 if the open space is forested and at zero otherwise, *PARK* is an indicator variable set at 1 if the open space is an urban park whose prime purpose is provision of wildlife habitat or dispersed recreation and that is characterized by predominantly native vegetation, and at zero otherwise, and *AG*, *PROT* and *PRIV* are indicator variables set at 1 if the open space is natural agricultural land (pasture, or pasture with some cropland), is protected, or is privately owned, respectively, and at zero otherwise.

Kroeger et al. found that the share of property values due to open space in the vicinity of a property (***%OSChange***) was highly significant. The elasticity of property value premiums with respect to the percentage of open space in the vicinity of a property is 0.42 while the coefficient on the open space percentage squared is -0.0068. Thus, an increase in the percentage of open space in an area from zero to ten percent will increase property values on average by 3.5 percent.[22] For forested, private, or protected open space or for natural area parks, this value is higher, while for agricultural open space it is lower. Because of the increasing power of the negative squared term for successively larger increases in open space, the marginal (i.e., additional) open space property premiums become negative once open space accounts for approximately 32 percent of the total area. This closely matches Walsh's results who found that in Wake county, North Carolina, marginal open space

[20] The remainder of the reviewed studies did not provide the required information for their inclusion in this analysis.
[21] The full model estimated by Kroeger et al. included a number of additional variables hypothesized to impact open space premiums. However, these were not found to be statistically significant and were excluded from the model used in this analysis.
[22] $0.4221*10 - 0.0068*(10^2) = 3.5$.

premiums turned negative for percentages of open space that exceed roughly one-third of the total area.

Kroeger et al.'s model explains nearly 50 percent of the variation observed in the data and as a whole is highly significant (p=0.0000). Their detailed results are shown in Table 4.2.

Table 4.2: Estimation results for the open space property premium model

Variable	*Unstandardized Coefficients*	*Std. Error*	*Standardized Coefficients*	*t-statistic*	*p-value*
(Constant)	-6.5903	1.6353		-4.0299	0.0002
%OSChange	0.4221	0.1290	1.3370	3.2714	0.0020
%OSChangeSq.	-0.0068	0.0032	-0.8801	-2.1432	0.0373
OS-Forest	2.7619	1.1329	0.3092	2.4379	0.0186
OS-Park	1.6768	1.9629	0.1073	0.8543	0.3973
OS-Grassland	-2.7367	1.1696	-0.2938	-2.3399	0.0236
Protected	3.5067	1.1039	0.3926	3.1767	0.0026
Private	5.3409	1.2818	0.6555	4.1667	0.0001
R^2		0.5433	N=55	F-statistic	7.9878
Adjusted R^2		0.4753		Prob.(F)	0.0000
Std. Error of the Estimate		2.9658			

Notes: OLS estimation. Dependent variable: %INCR_PV. Wetland is set as the baseline land cover to estimate the coefficients of the other land covers, and thus is not listed under the land cover proxies. The model can be re-parameterized to set any other land value as the base land cover without affecting the open space premium estimates.

Source: Kroeger et al. (2008)

It should be noted that this model likely overestimates the attenuation of the size of marginal open space premiums that results from large open spaces, for reasons explained in detail in Kroeger et al. (2008). As a result, the model is likely to underestimate premiums in areas with large amounts of open space.

Even though no study is available on the open space value in our study area, Bin and Polasky's (2005) study context is reasonably similar to our study area. Specifically, wetlands account for a high share (55 percent) of the total area in our five-county area, as they did in Bin and Polasky's study. Thus, as in their study area, wetlands are not a scarce resource in our area, and it is likely that their property value impacts in our area are similar to those estimated by those authors for Carteret County.

Kroeger et al.'s (2008) open space premium model was estimated based on the property premium values reported in the literature. Due to the many positive property value impacts reported in wetland studies, the coefficient on the wetland variable in the model is positive. Thus, the model generates positive open space premium values for wetlands. In order to avoid overestimation of open space values in our study area, we excluded wetlands from the open space value estimation and only estimate open space premiums for forests and grasslands.

Methods used in open space premium estimation

Kroeger et al.'s (2008) property value premium model function (eq. 1) was used to estimate open space premiums for properties in the study area. We define open space as undeveloped, relatively undisturbed natural land. We excluded areas of open water from the analysis, since the premium model does not include this land cover type as an open space land cover option. In addition, as discussed above, we excluded wetlands from the analysis.

Application of the open space premium model requires information on the amount of open space in each area of analysis, measured as percentage within a one-mile radius around each town or settlement, the type of open space (forest, grassland), and the number and market value (as approximated by assessment value) of the properties in the area. We obtained this information from aerial photographs of the landscape (orthophotos), tax parcel shapefiles, and the 2001 National Land Cover Database (NLCD).

For each county we used compressed county mosaics (CCMs). CCMs are compressed digital orthophotos[23] formed into a single mosaic of the county. CCMs were released in 2008 for Beaufort county, and in 2009 for the four other counties. The orthophotos were obtained from the National Agriculture Imagery Program (NAIP) which makes digital orthophotography available to governmental agencies and the public.[24] All the imagery acquired from NAIP represents agricultural growing seasons. Because the land cover types within the NLCD are nine years old this "leaf-on" imagery provided a useful comparison between the NLCD land cover types and the actual landscape. In addition, the orthophotos allowed us to determine where the densest populated areas were for each town within a one mile radius of the study area.

To determine the open space area for each town, we converted features (a shapefile)[25] that identified the towns in each county to graphics. We marked our study area for each town by moving the graphics to the densest areas without open water. Once in place, the graphics were converted back to a shapefile, and using the ArcGIS buffer tool, a one mile radius was drawn around each feature.

Once we determined the individual open space analysis areas, we overlaid the tax parcel data from each county and used the subset of parcels that fell within each town's study area. A table was generated that contained the total number of single family homes, the tax appraised value for each home, and whether or not it was public or private housing or buildings. The level of detail and accuracy varied between each county. Beaufort and Hyde counties did not include building definitions or zoning which made it difficult to determine which properties within the study area contained single family residences. For these counties we eliminated properties listed by commercial, church, or government names and included only properties with a first and last name. For all counties we eliminated any building values below $10,000, vacant lots, or properties without a

[23] An orthophoto is an aerial photograph that has been corrected for topographic relief, lens distortion, and camera tilt through a process called orthorectification. This process allows for complex spatial analysis because distance and area are uniform in relationship to real world measurements.

[24] NAIP is administered by the USDA's Farm Service Agency (FSA) through the Aerial Photography Field Office in Salt Lake City. Their aerial imagery is used as a base layer for GIS programs in the USDA FSA's County Service Centers, and is used to maintain the Common Land Unit (CLU) boundaries.

[25] A shapefile is a spatially represented digitized shape called a feature. They do not store topology and can contain only one feature class. Features may be points, lines, or polygons and their attribute data are stored as dBase files linked with a collection of files with a common name (e.g. roads.shp, roads.dbf, roads.shx).

significant structure (Table 4.3). The mean property value was calculated from the remaining parcels in the table.

Table 4.3: Parcel data sources used in the analysis

County	*Data*	*Data Notes*	*Orthophoto year*
Beaufort	2009 tax parcels do not include zoning or building definition	Removed all parcels that had building values less than $10,000 and names that were obvious commercial, church, or public property	2008
Hyde	2009 tax parcels do not include zoning and could not match up building definitions with spatial data	Removed all parcels that had building values less than $10,000 and names that were obvious commercial, church, or public property	2009
Tyrell	2005 tax parcels with building definitions	Included all parcels with a building class "dwelling" and building values greater than $10,000	2009
Washington	2009 tax parcels with zoning and building definitions	Included all parcels with a type use of "1" which means single family residence and all building values greater than $10,000	2009
Dare	2009 tax parcels with zoning	Included all parcels with a zoning of single family residence and all building values greater than $10,000	2009

It is worth noting that the use of tax assessment values may result in biased estimates of home values if values have changed since the last assessment. However, comparison with data from the 2000 National Census revealed that property values in the area remained fairly constant between 2000 and 2009.

To determine the percentage and type of land cover within the individual open space analysis areas we converted the shapefile of buffered features into individual shapefiles. We did this by using a custom-written Python script.[26] Once we had individual shapefiles we clipped the NLCD land cover raster[27] to each town, added it to the display, and used the ArcGIS Zoom to Layer function for quick analyses of the study area. To make this process more efficient, we used a custom-written model that extracted the land cover data for each town and exported it into a spreadsheet with different land cover types and the number of raster cells by land cover type. Acreages were calculated by multiplying the number of raster pixel cells by the real world area (900 square meters) and converting into acres.[28]

To determine land cover types within each study area we used the 2001 NLCD compiled across all 50 states and Puerto Rico as a cooperative mapping effort of the Multi-Resolution Land Characteristics 2001 Consortium (Homer et al. 2004). This land cover database was created using mapping zones and contains 28 standardized land cover types. Acreage totals for forest, grassland, pastureland, and wetlands were calculated to determine the total open space within the study area.

[26] Natalie Dubois, Conservation Planning Associate, Defenders of Wildlife.

[27] A raster is a spatially represented file that defines the geographic surface as rows and columns of pixels. For the NLCD land cover raster each pixel cell represents a real world area of 900 meters squared of a defined land cover type (e.g. wetland or forest).

[28] 50 raster pixel cells of wetland x 900 square meters x 0.000247105381 acres = 11.12 acres

The land cover type of shrub/scrub was compared with the NLCD data and the orthophoto. If the orthophoto verified the existence of forest with this land cover type then it was included, if it looked like cropland, or barren land then it was excluded from the analysis. Land cover types that comprised more than 20 percent of the total open space within the study area were included in the open space analysis. Open water and developed land were excluded. Land ownership was identified by zooming to the town layer and applying the ArcGIS Information tool. For application of the open space premium model, we only included a land ownership type (private, public) if it accounted for at least 20% of the area within a 1-mile radius of the town.

With the open space percentage (*%OSChange* in eq.1) thus identified for each subsection of our study area, we set the indicator variables in the function at their appropriate values. If a particular open space was in forest cover, the *FOR* variable was set to (1) If it was a grassland, the *Grassland* variable was set to 1. When both forest and grassland were present (i.e., accounted for more than 20 percent each of the total open space in an area), we ran the model twice, once with forest set as the land cover type and once with grassland set as the cover type, and took a weighted average of the resulting open space premium percentages.

Results: Estimated open space premiums captured in residential property values in Northeastern North Carolina

Table 4.4 presents the aggregate open space premiums for each of the five counties in our study area. These results show that in 2009, the total residential property value premium in our study area attributable to forests and grasslands was estimated at approximately $50 million (2009$). More detailed information on premiums in each of the residential areas is found in Appendix 3, Table A3.1.

Table 4.4: Total and average estimated residential open space premiums in study area counties

County	*Number of properties included in analysis*	*Total OS premium in county (2009$)*	*Average premium per property (2009$)*
Hyde	519	1,516,861	2,923
Beaufort	7,547	40,240,565	5,332
Dare	0 (all wetlands)	-	-
Tyrell	336	2,092,046	6,226
Washington	1,536	6,706,284	4,366
TOTAL	9,938	50,555,755	5,087

Our results indicate that the estimated open space premium received by residential properties range from less than one (0.4) percent to 8 percent of property value (Table APP-A). This range is a result of the different quantity, land cover type and protection status of the open spaces found in the vicinity of the residential areas. The size of the total open space premium received in each county ranges considerably as a function of the number and value of properties in each county, from an estimated $1.5 million in Hyde County, to an estimated 40 million in Beaufort County (Table 4.4). Since open space premiums are reflected in property values, they form part of the assessed property value. With average property tax rates in the four counties included in our estimation (with Dare County excluded because all open spaces were wetlands; see discussion above) of around one percent (North Carolina Department of Revenue, 2009), the open space in the study area contributes an estimated $500,000 per year to property tax revenues.

It bears emphasizing that the open space value of natural lands is only a portion of the total economic value these lands generate, namely, the amenity value homeowners place on scenic views and on easy access to green areas. Natural lands provide a multitude of additional benefits, from habitat for fishable, huntable and viewable wildlife and threatened, endangered and rare species, to water quality and carbon sequestration, to name but a few. This is also true for wetlands (Bin and Polasky, 2005), which we assumed did not generate property value premiums because of the findings of another study carried out in the same region of the state (ibid.).

V. Conclusions and Policy Recommendations

Although attempts to reestablish red wolves in the wild began on the Alligator River National Wildlife Refuge, the 100 to 120 individuals that currently make up the wild red wolf population has spread across five Northeastern North Carolina counties. The wolves benefit from ongoing efforts to restore ecosystem services in North Carolina. For example, 66,000 acres of formerly drained pocosin wetlands on peat soils are being restored to their former wetland forest habitat on Pocosin Lakes NWR and Alligator River NWR. This restoration primarily focuses on ending massive release of greenhouse gases which has occurred after drainage and clearing of these peat soils for agriculture and other uses, but it also contributes to improved red wolf habitat.

Red wolves benefit as well from various options for purchasing ecosystem services, such as the Conservation Reserve Program's payments to restore riparian areas from crop uses to forest uses to reduce nutrient pollution in the Pamlico Sound. Although this program provides payments comparable to farmers' income from cropping the land, it only alters a small percentage of land use within the study area.

Red wolves potentially could benefit, also, from payments for forest related open space services that increase value of adjacent homes. Open space premiums average $5,000 per property, and the total property value in the study area resulting from with open space is $50 million. Analysis suggests that shifting cropland to forest uses in the five county study areas could benefit property values, water quality, protection of at risk wildlife species, carbon sequestration, and red wolf habitat. However, creating markets to capture these benefits remains challenging.

Analysis found that payments for carbon sequestration services offer the greatest potential to alter Eastern North Carolina's rural landscapes in ways that favor red wolves. International carbon prices at of around $30 per ton CO_2e would be sufficient to cause farmers to convert substantial acreage of local cropland to forest uses and might in some cases compete with forest harvest for timber. Thus, carbon markets could both encourage farmers to convert cropland to forest and might encourage more preservation of some existing forests. This would alter the five counties' landscape in ways that substantially increase forest habitat favored by red wolves. However, attempts to bring the U.S. into international carbon markets so far have been unsuccessful. In the absence of a U.S. cap and trade program, carbon prices in the $5 per ton range are likely to continue. This price is too low to substantially affect forest land use in the study area.

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VII. Appendices

Appendix 1: Land Use Information from 2008 Ecosystem Services and Markets Farm Operator Survey
Source: Kramer and Jenkins, 2009.

Table A1.1: Land management

	% HH Income from land	Acres in crop production	Primary use of land	Highest value commodities produced	% w/ acres in permanent easement
Question #	7	8	9	10	11
Stat	46% (mean)	777 (mean) 180 (median)	84% agriculture	Corn 35% Soybeans 16% Cotton 14%	7% (mean)

Table A1.2: Conservation program participation, past, current, and potential

	Past participation in conservation program	*Current* participation in conservation program	Consider participating in PES	Q33. Importance of program attributes (scale of 1 to 5)		
Question #	22	23	30	Contract length	Program administration	Payment level
Stat	51% YES 46% NO	33% YES 64% NO	63% YES 7% NO 30% Don't know	4.14 (mean)	3.81 (mean)	4.33 (mean)

Figure A1.1: Reasons respondents chose not to enroll in a conservation program (Q24)

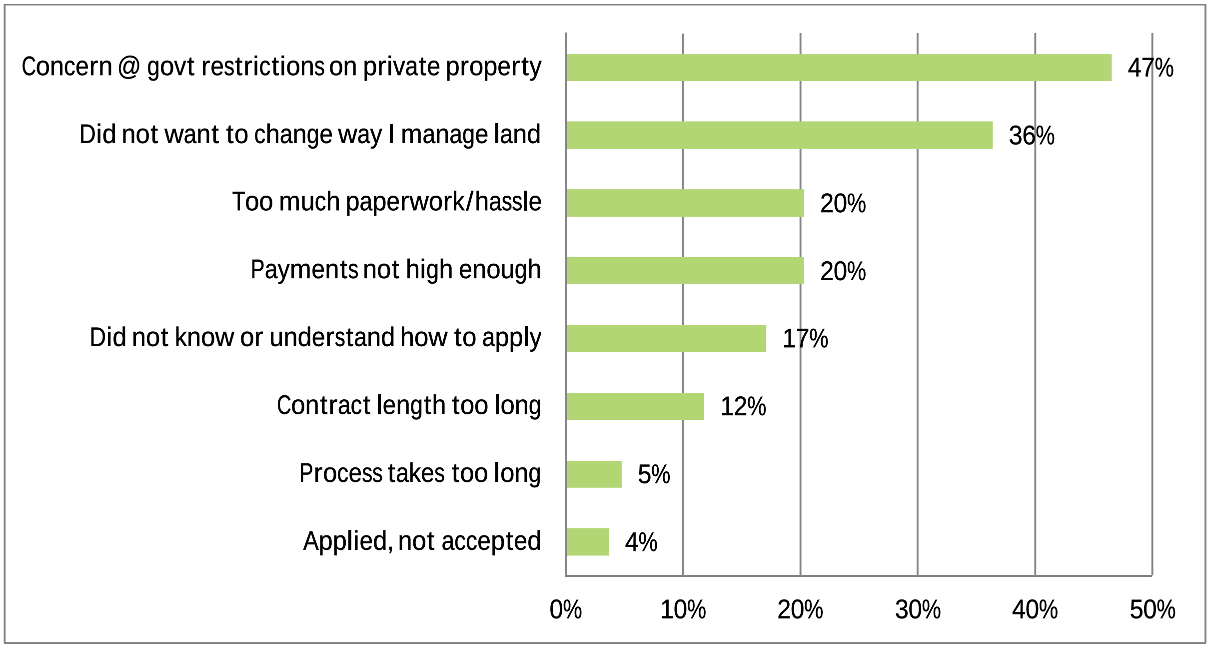

Table A1.3: Share of different land uses of total acres

	Avg	Bertie	Martin\Pitt	Beaufort	Washington	Hyde	Tyrrell	Dare
N	269	78	7	67	34	31	29	3
Crop prod	69.2%	57.4%	57.5%	71.9%	77.0%	74.3%	77.3%	76.7%
Livestock	2.1%	2.8%	0.0%	1.8%	1.2%	0.3%	2.4%	0.0%
Planted forest	15.1%	24.1%	26.8%	14.5%	8.3%	10.3%	5.2%	18.3%
Natural forest	12.6%	15.4%	15.7%	10.8%	13.2%	9.6%	15.1%	5.0%
Marsh	1.1%	0.4%	0.0%	0.9%	0.3%	5.5%	0.1%	0.0%

Table A1.4: Acres in different land uses (mean values)

	Avg	Bertie	Martin/Pitt	Beaufort	Washington	Hyde	Tyrrell	Dare
N	269	78	7	67	34	31	29	3
Crop prod	658	307	495	579	651	1317	1161	1001
Livestock	20	15	0	14	10	5	36	0
Planted forest	144	129	231	117	70	183	78	239
Natural forest	120	82	135	87	112	170	227	65
Marsh	10	2	0	7	3	98	2	0
Total acres	952	536	861	804	845	1773	1504	1305
Perm easement	50	1	100	3	3	221	17	0
Conserv program	70	53	21	63	68	141	50	246

Table A1.5: Percentage of farms by primary use and most valuable commodity crop

	Avg	Bertie	Martin	Beaufort	Washington	Hyde	Tyrrell
N	269	78	4	71	34	31	29
Primary use - Ag	83.7%	78.3%	75.0%	83.1%	86.1%	87.5%	93.3%
Primary use - Timber	5.0%	10.8%	0.0%	2.8%	2.8%	0.0%	3.3%
Primary use - Ag & Timber	6.7%	4.8%	25.0%	9.9%	5.6%	9.4%	3.3%
Top crop - Corn	31.1%	9.3%	0.0%	38.1%	28.1%	53.3%	62.1%
Top crop - Cotton	13.6%	17.3%	100.0%	12.7%	6.3%	20.0%	3.5%
Top crop - Soy	13.2%	12.0%	0.0%	9.5%	28.1%	10.0%	10.3%
Top crop - Timber	11.7%	20.0%	0.0%	9.5%	3.1%	0.0%	10.3%
Top crop - Tobacco	6.6%	8.0%	0.0%	12.7%	3.1%	0.0%	0.0%
Top crop - Broilers	4.3%	14.7%	0.0%	0.0%	0.0%	0.0%	0.0%
Top crop - peanuts	3.1%	8.0%	0.0%	1.6%	0.0%	0.0%	0.0%

Appendix 2: Estimates of Carbon Storage for Afforestation

Source: USFS 1605(b) tables for Southeast Region

Table A2.1: Farm A

Afforested **Longleaf-slash pine**

Time	Total non-soil carbon (USFS tables)	Incremental gain in total non-soil C	Incremental gain in total non-soil C	Soil C (USFS tables)	Incremental gain in soil	Incremental gain in soil C	Total incremental C	Total incremental C
Yrs	tC/ha	tC/ha	tCO2/ha	tC/ha	tC/ha	tCO2/ha	tCO2/ha	tCO2/ac
0	4.2	0	0.0	82.5	0	0.0	0.0	0.0
5	13.6	9.4	34.5	82.8	0.3	1.1	35.6	14.4
10	25.4	11.8	43.3	83.6	0.8	2.9	46.2	18.7
15	34.9	9.5	34.8	84.9	1.3	4.8	39.6	16.0
20	46	11.1	40.7	86.6	1.7	6.2	46.9	19.0
25	56.6	10.6	38.9	88.6	2	7.3	46.2	18.7
30	66.1	9.5	34.8	90.9	2.3	8.4	43.3	17.5
35	75.1	9	33.0	93.2	2.3	8.4	41.4	16.8
40	83.4	8.3	30.4	95.5	2.3	8.4	38.9	15.7
45	91.1	7.7	28.2	97.8	2.3	8.4	36.7	14.8
50	98.4	7.3	26.8	99.9	2.1	7.7	34.5	13.9
55	104.8	6.4	23.5	101.8	1.9	7.0	30.4	12.3
60	110.6	5.8	21.3	103.5	1.7	6.2	27.5	11.1
65	116.1	5.5	20.2	105	1.5	5.5	25.7	10.4
70	121.4	5.3	19.4	106.2	1.2	4.4	23.8	9.6
75	125.8	4.4	16.1	107.1	0.9	3.3	19.4	7.9
80	130.3	4.5	16.5	107.9	0.8	2.9	19.4	7.9
85	134.9	4.6	16.9	108.5	0.6	2.2	19.1	7.7
90	138.5	3.6	13.2	109	0.5	1.8	15.0	6.1

Annualized tCO2/acre values:		
	1-5 yrs	2.9
	6-10 yrs	3.7
	11-15 yrs	3.2
	16-20 yrs	3.8
	21-25 yrs	3.7
	26-30 yrs	3.5
	31-35 yrs	3.4
	36-40 yrs	3.1

Table A2.2: Farm B

Afforested	Oak-gum-cypress							
Time	Total non-soil carbon using USFS tables	Incremental gain in total non-soil C	Incremental gain in total non-soil C	Soil C using USFS tables	Incremental gain in soil C (USFS tables)	Incremental gain in soil C (USFS tables)	Total incremental C	Total incremental C
Yrs	tC/ha	tC/ha	tCO2/ha	tC/ha	tC/ha	tCO2/ha	tCO2/ha	tCO2/ac
0	1.8	0	0.0	118.5	0	0.0	0.0	0.0
5	10.9	9.1	33.4	118.9	0.4	1.5	34.8	14.1
10	25.8	14.9	54.6	120.1	1.2	4.4	59.0	23.9
15	37.2	11.4	41.8	121.9	1.8	6.6	48.4	19.6
20	48.6	11.4	41.8	124.4	2.5	9.2	51.0	20.6
25	58.9	10.3	37.8	127.2	2.8	10.3	48.0	19.4
30	67.5	8.6	31.5	130.5	3.3	12.1	43.6	17.7
35	77	9.5	34.8	133.8	3.3	12.1	46.9	19.0
40	88	11	40.3	137.2	3.4	12.5	52.8	21.4
45	98.2	10.2	37.4	140.4	3.2	11.7	49.1	19.9
50	107.4	9.2	33.7	143.5	3.1	11.4	45.1	18.3
55	116.7	9.3	34.1	146.2	2.7	9.9	44.0	17.8
60	126.4	9.7	35.6	148.7	2.5	9.2	44.7	18.1
65	136.1	9.7	35.6	150.7	2	7.3	42.9	17.4
70	145	8.9	32.6	152.4	1.7	6.2	38.9	15.7
75	152.6	7.6	27.9	153.8	1.4	5.1	33.0	13.4
80	159.4	6.8	24.9	155	1.2	4.4	29.3	11.9
85	167.8	8.4	30.8	155.8	0.8	2.9	33.7	13.7
90	177	9.2	33.7	156.5	0.7	2.6	36.3	14.7

Annualized tCO2/acre values:

1-5 yrs	2.8
6-10 yrs	4.8
11-15 yrs	3.9
16-20 yrs	4.1
21-25 yrs	3.9
26-30 yrs	3.5
31-35 yrs	3.8
36-40 yrs	4.3

Appendix 3

Table A3.1: Towns in study area for which open space premiums were estimated

County	*Town*	*Number of housing units*	*OS as % of area within 1 mile of avg. property*	*Median home value in 2009*	*Avg. property premium (% of property value)**	*Total value (million 2009$)*
Hyde	Ponzer	33	26%	$111,528	7.9	$289,482
	Mount Olive	36	38%	$111,973	7.7	$308,616
	Beulah	34	17%	$85,523	6.7	$193,883
	Swanquarter	97	23%	$76,715	4.9	$360,837
	Fairfield	79	25%	$74,094	2.3	$132,028
	Slocum	22	15%	$107,257	0.8	$19,169
	Nebraska	29	14%	$86,380	0.5	$13,622
	Middletown	68	15%	$78,944	0.8	$41,279
	Engelhard	121	17%	$101,104	1.3	$157,945
Beaufort	South Creek	60	48%	$65,017	6.1	$238,421
	Edward	122	60%	$49,248	2.8	$167,614
	Reads Chapel	47	41%	$57,190	4.7	$124,701
	Aurora	260	26%	$55,189	5.2	$734,602
	Campbell Creek	36	48%	$56,687	6.0	$121,644
	Small	102	72%	$56,503	4.2	$239,643
	Batts Crossroads	46	63%	$76,786	5.5	$195,647
	Pike Road	83	19%	$73,433	4.3	$257,531
	Leggetts Crossroads	12	68%	$120,394	1.9	$27,055
	Stallings Crossroads	51	55%	$87,146	4.2	$184,804
	Old Ford	96	33%	$80,096	8.0	$614,156
	Swindell	38	13%	$106,314	0.4	$15,277
	Pinetown	136	38%	$68,043	7.6	$707,537
	Terra Ceia	70	16%	$138,575	1.1	$105,852
	Latham	187	35%	$132,623	5.2	$1,287,603
	Pantego	225	24%	$57,126	7.7	$994,411
	Rosedale	653	32%	$106,471	5.3	$3,670,566
	Leechville	28	59%	$70,429	5.5	$109,230
	Pineygrove	249	47%	$79,214	6.3	$1,246,991
	Five Points	69	70%	$87,066	5.5	$332,760
	Alligoods	182	61%	$74,598	5.5	$752,024
	Washington	1862	17%	$92,678	5.9	$10,118,256
	Rodmans Quarter	72	31%	$59,289	8.0	$342,777
	Boyds Fork	73	70%	$71,149	5.5	$287,690
	Bellhaven	427	20%	$56,790	4.5	$1,078,785
	Bunyan	229	28%	$91,753	8.0	$1,677,856
	Hootentown	820	37%	$107,487	7.8	$6,860,569
	Chocowinty	267	33%	$71,594	8.0	$1,530,546
	Midway	87	69%	$75,361	5.5	$363,161
	Whitepost	54	42%	$85,861	4.5	$204,161
	Bath	270	36%	$172,871	7.8	$3,656,949
	Hackney	145	32%	$67,843	2.5	$248,403
	Winsteadville	24	40%	$74,190	7.5	$132,923

- Over -

- continued -						
County	*Town*	*Number of housing units*	*OS as % of area within 1 mile of avg. property*	*Median home value in 2009*	*Avg. property premium (% of property value)**	*Total value (million 2009$)*
	McConnell	50	65%	$56,620	5.5	$156,810
	Gilead	92	84%	$83,294	2.8	$213,778
	Ransomville	59	51%	$75,516	2.8	$185,261
	Gaylord	71	32%	$89,776	8.0	$511,010
	Rover	30	93%	$126,647	2.8	$105,993
	Wilmar	54	38%	$75,254	5.0	$201,563
	Coxs Crossroads	55	84%	$57,421	2.8	$88,104
	Bonnerton	14	63%	$43,991	5.5	$34,113
	Stilley	4	72%	$49,034	5.5	$10,864
Washington	Beasley	12	68%	$156,263	5.5	$103,865
	Blount	17	38%	$118,120	7.6	$153,429
	Creswell	109	30%	$62,574	8.0	$547,632
	DavenportForks	33	32%	$77,968	8.0	$206,497
	Hinson	17	90%	$82,141	5.5	$77,347
	Hoke	23	69%	$80,002	5.5	$101,921
	Pineridge	93	36%	$94,521	5.1	$446,328
	Plymouth	739	23%	$51,808	7.6	$2,913,239
	Ren	106	70%	$73,954	5.5	$434,211
	Roper	295	22%	$58,051	7.5	$1,286,679
	Wenona	30	13%	$84,632	3.1	$78,482
	Westover	62	31%	$108,913	5.3	$356,654
Tyrell	Columbia	177	63%	$96,068	5.5	$941,857
	Dillionridge	1	43%	$56,446	4.6	$3,127
	Fryingpan	1	23%	$46,928	2.0	$962
	GalileeMission	4	18%	$212,958	1.4	$12,062
	Gumneck	40	22%	$79,883	4.8	$151,076
	Millsridge	1	50%	$188,421	1.9	$2,061
	Newfoundland	14	60%	$67,563	5.5	$52,393
	Riverneck	26	31%	$181,747	8.0	$379,468
	Soundside	50	45%	$121,380	6.7	$407,572
	Woodley	22	20%	$88,787	7.2	$141,468
						$50,555,755

* Estimated based on the specific open space type(s) (forest, grassland) found in the area.

Note: In cases where open space accounted for more than 50 percent of total area, the value was set to 50 percent as that value was the upper end of open space share in the studies used to estimate the model.