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Regional carbon dynamics in the southeastern U.S. coastal plain: Balancing land cover type, timber harvesting, fire, and environmental variation

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[1] Understanding regional carbon budgets is a leading issue in carbon cycling research, but issues of measurement difficulty, scale, boundaries, and logistics compromise estimates at areas larger than stands or research plots. We studied four 15 $\times$ 15 km sample areas to examine land management and wildfire effects on carbon storage dynamics in the forested southeastern U.S. coastal plain region from 1975 to 2001. Carbon exchange and storage rates were estimated using satellite remote-sensing methods coupled with micrometeorological and biomass measurements. Carbon losses occurred by timber harvesting and fire, and carbon release continued for four years following clearing, suppressing landscape carbon gain proportional to the cleared area. Carbon accumulated at an average rate of 90,000 t C yr^{-1} in the landscape (total area 900 km^2) from 1975–2000, or ~1 t C ha^{-1} yr^{-1}. Interannual variation was related mainly to the magnitude of annual plantation timber harvesting. Wildfires were rare and their effects on carbon balances consequently small, despite having large local impact. Previous studies in the area demonstrated that environmental fluctuations had little direct effect on the net landscape exchange of carbon, although indirect effects included higher probability of fire during droughts and shifts in harvesting to drier sites during wet periods. Although this study was a simple aggregation of carbon cycle components from fine spatial scale (Landsat images) to the landscape, the extrapolation incorporated important spatial and temporal environmental heterogeneities and led to the unexpected suggestion that the industrial forests of the southeast U.S. Coastal plain are a long-term carbon sink. The analysis also revealed specific uncertainties in our scaling efforts that point to future research needs.

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1. Introduction

[2] The carbon budget of ecosystems is fundamentally important from the perspectives of both ecology and economics. Ecologists have long sought reliable measures of ecosystem level budgets so that limits can be placed on the rates of the biological processes of fixation and release that are very difficult to measure and scale, and because the balances of other ecosystem fluxes (e.g., retention of nitrogen) are related [*Gorham et al.*, 1979]. The interest of economists derives both from the business of forestry, and from the recent emergence of carbon as a potentially tradable global commodity [*Sandor and Walsh*, 2001; *Stainback and Alavalapati*, 2002].

[3] The appropriate scale for measurement of an "ecosystem balance" is also a fundamental issue, not only because boundaries of ecosystems are often set arbitrarily [*Allen and Hoekstra*, 1990], but also because conclusions regarding carbon balance are dependent on the temporal and spatial scales of both controlling factors and of observation [*Harmon*, 2001]. If the system to be quantified is very small (most ecological measurements at time steps less than a day are made to characterize areas < 100 m^2 in extent: [*Porter et al.*, 2005]), then many such areas may need to be studied to ensure that plot-plot heterogeneity is adequately accounted for. Larger areas are inherently difficult to quantify because of local heterogeneities in soil, topography and land use. The most common method now used for measurement of carbon fluxes, eddy covariance from a tower above the study ecosystem, has a variable and highly uncertain plot size, or footprint. The most common models used for estimating "footprints" of the eddy covariance measurements [*Scheupp et al.*, 1990; *Schmid*, 1997, 2002], are

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complicated and do not model heterogeneous landscapes well. In addition, high expense limits the numbers of flux towers and encourages their placement in the middle of reserves and other attractive study areas that may not be representative of the surrounding landscape. Terminology is also confusing when large-scale carbon budgets are constructed. The terms net biome exchange and net biome production have been introduced to characterize the carbon balance of an area large enough to contain a number of distinct ecosystems or land cover types, and where, at the same time, the full range of potential disturbances is enabled [*Schulze and Heimann*, 1998]. It is at these scales where comparisons can be made with the outputs of most climate models and therefore where attention is being increasingly focused. Additionally, *Houghton et al.* [2001] suggest that a combination of ground based allometric estimates along with satellite imagery would provide a good indicator of large-scale biomass estimates in a more cost effective manner.

[4] The timescale of measurement is also critical. The focus of most tower studies is on understanding interannual fluctuations of small, intact ecosystems, primarily relating them to changing environmental conditions [e.g., *Hui et al.*, 2003]. The longest-running single study (at Harvard Forest, Massachusetts) is still less than 15 years old and most data sets are less than 5 years in length [*Olson et al.*, 2004]. At longer timescales, disturbances from fire, logging, or land use/land cover change become increasingly relevant and must at some point become incorporated for estimates of large-scale carbon exchange to be realistic. It will only be through a combination of multiple measurement locations (over a large spatial scale), made over long time periods, supported by reconstruction of historic patterns, and including disturbances, that an understanding of trends in regional carbon dynamics can be obtained.

[5] This study took place in the lower coastal plain of the southeastern U.S. east of the Mississippi River, which covers more land area (nearly 600,000 km^2) than Germany, the Benelux nations and the United Kingdom combined. It has a large and rapidly growing human population, while a large proportion of the landscape is intensely managed for silviculture and agriculture. The region also contains a high diversity of natural ecosystems, related mainly to subtle variations in topography, geology, and soils within a generally uniform and moderate subtropical climate [*Myers and Ewel*, 1990; *Sampson*, 2004]. Our objective was to estimate the carbon balance of a landscape within this region over the past quarter century (1975–2001), by quantifying the major factors controlling the balance, including timber harvesting, changes in land use and land ownership, environmental conditions, and wildfire. We accomplish this by linking landscape-wide estimates of forest age, determined by examination of sequential, multitemporal Landsat scenes, with eddy covariance and biomass measurements along plantation chronosequences and in natural ecosystems. We hypothesized that the net regional carbon budget would balance (average 0) over this period, which would occur if vegetation regrowth and net ecosystem carbon gain equaled losses of carbon from timber harvesting and fire, with the effects of interannual environmental fluctuations averaging over time, which is reasonable given that there is no discernable long-term trend in precipitation and stream flow over the time period of the study [*Zorn and Waylen*, 1997].

2. Study Area

[6] The study focused on nonpublic (private industrial, private nonindustrial) lands in the north central Florida portion of the region (Figure 1). Four sample areas, each approximately 15 $\times$ 15 km, were selected within the footprint of Landsat WRS II path 17/row 39. A single Landsat scene was used to avoid the complications of normalizing different images over space as well as time. We refer to the physical scale of our results, where we present averages or aggregations over the four sample areas, as the "landscape." The sample areas also would characterize the lower coastal plain region to the degree that our landscape is representative (which we did not test). Study areas were located so that they fell completely within the boundaries of single counties, avoided publicly owned lands (e.g., parks, military, protected areas), and consisted mostly of forests. One area in Alachua County was designated as a study area because it included locations of current and previous research [e.g., *Gholz and Fisher*, 1982; *Powell et al.*, 2005]. Three other areas, in Clay, Hamilton, and Union Counties, met all the criteria. The actual study area boundaries coincided with the Public Land Survey System (PLSS) boundaries that are used for the division of land in Florida because our project also studied the influence of land ownership on ecosystem processes [see *Barnes et al.*, 2003]. The PLSS is based on surveyed townships and ranges, each of which is 6 $\times$ 6 mi. To approximate the 15-km square, we used a 9 $\times$ 9 mi area, which is 14.481 km on a side. Each area varied slightly from this size because of minor deviations in the land survey caused by the need to conform to certain landscape features, as well as land ownership that existed prior to the imposition of the PLSS.

[7] The dominant upland ecosystems in the region are pine flatwoods, with mixed hardwoods and pines at intermediate moisture levels, and bottomland hardwood forests near streams and rivers. Cypress (*Taxodium ascendans*) wetlands dot the uplands in areas of high water table or depressions caused by buried sinkholes in the limestone bedrock [*Myers*, 1990]. Only 1.5% of the area of the original, upland *Pinus palustris* (longleaf pine) forest remains after clearing during the past 150 years, although they once covered 25 million ha [*Myers*, 1990]. Widely spaced trees and frequent, low-intensity ground fires characterized these original forests. Most of the natural forests in the region today are denser, mixed pine stands, naturally regenerated after abandonment and are often long unburned. Many of the flatter areas of the region were converted to plantations of the native slash pine (*Pinus elliottii* var. *elliottii*) in the 1950s and 1960s. The total area in plantations has been roughly constant for the past 20 years, while the total forest area of the region has declined, primarily because of conversion from forest to developed urban use [*Wear*, 2002]. The intensity of forest management is increasing, with rotations (the period between planting and harvest) becoming shorter (now about 18–20 years on average). Soil tillage is commonplace at stand establishment, herbicides are routinely used early in stand development, fertilization is now standard, and the use of

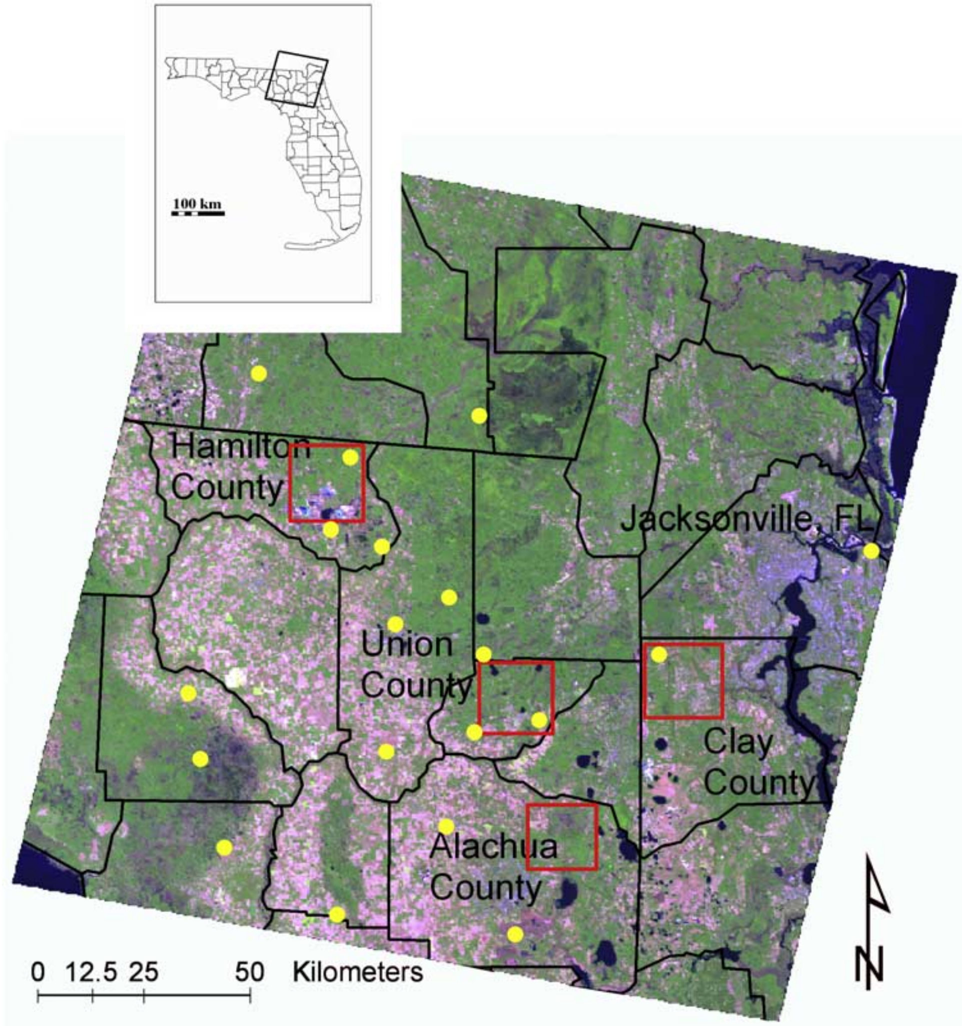

Figure 1. Composite (bands 5,4, and 3 = RGB) image of Landsat WRS 2 path 17 row 39 scene showing the study region, random points (yellow dots), and final sample areas (red boxes).

genetically improved growing stock is widespread [*Fox et al.*, 2004]. Conversely, much of the nonindustrial private and state land is managed less intensively to restore more natural forest conditions. Thus a highly contrasting mixture of management strategies, from intensely active to passive, has emerged across this landscape over time.

[8] Annual precipitation and streamflow of the region are strongly influenced by El Nino-Southern Oscillation cycles [*Zorn and Waylen*, 1997], with wet El Niño and dry La Niña years that can cause flood years followed by drought years. Both hurricane landfall and severe droughts occur about every 15 years on average [*Gholz and Boring*, 1991], with the latter often leading to severe wildfire conditions. Elevations range from near sea level in Clay County to the high point of 63 m in Hamilton Co. Topography is very gently rolling with slopes > 5% very rare. The surface topography is controlled by buried Karst, with a thick sand mantle overlying a confining layer over Tertiary limestone bedrock. Soils are mostly excessively drained Entisols and poorly drained Spodosols (with spodic and some argillic horizons) [*Brown et al.*, 1990]. Hydric soils are spotted throughout the region. Despite the calcareous bedrock, surface waters are acidic and soils infertile.

3. Methods

[9] Satellite imagery provides the only means for mapping regional ecosystem properties with fine-grained spatial and temporal resolution for the past 30 years. We selected four critical times to account for phenological variation (December/January: end of litterfall; early March: minimum leaf area index or LAI; June: middle of leaf area expansion; and September/October: Maximum LAI) and acquired all the available, cloud-free (in the study areas) Landsat MSS, TM and ETM+ scenes for Path 17, Row 39 from 1972 to 2001. There was a coverage gap between 01/23/76 and 03/27/81 (four missing years) because of cloudiness or technically damaged scenes. Other than those years, at least one acceptable scene was available from December to March for every year of the study. Complete phenological coverage (all four periods) was achieved in only two years (1988–1989 and 1997–1998).

Table 1. Look-up Table for Landscape Estimates of NEE and C Losses[a]

Age Class (Years Since Clearing)	NEE, T C ha^{-1} yr^{-1}	Losses to Cutting, T C ha^{-1} yr^{-1}	Loss to Fire, T C ha^{-1} yr^{-1}
Pine Plantations			
*0	−12.5	0	0.1
*1	−9.1	0	0.6
*2	−4.7	0	1.4
*3	−0.2	0	1.9
*4	0.9	0	2.3
5	1.9	0	2.9
6	2.9	0	3.0
7	3.9	0	3.2
8	4.9	6.8	10.2
*9	5.9	9.4	13.1
*10	6.5	12.1	16.1
*11	6.3	14.8	19.0
*12	6.8	17.4	22.0
13	6.3	20.1	24.9
14	6.5	22.7	27.9
15	6.5	25.4	30.8
16	6.5	28.0	33.8
17	6.5	30.7	36.7
18	6.5	33.4	39.6
19	6.5	36.0	42.6
20	6.5	38.7	45.5
21	6.5	41.3	48.5
22	6.5	44.0	51.4
*23	6.5	46.6	54.4
*24	7.5	49.3	57.3
*25	6.4	52.0	60.3
**Natural Mixed Pine*			
	1.8	1.7	2.0
**Cypress/Wetlands*			
	0.6	5.4	6.4

[a]Net ecosystem exchange (NEE) data (asterisk indicates years for which there were measurements: NEE from years without measurements were estimated with linear interpolation) are from *Clark et al.* [1999, 2004], *Powell* [2002], and *Powell et al.* [2005]; Carbon loss data are from *Gholz and Fisher* [1982] and *Clark et al.* [1999, 2004]. We assumed that there are no timber removals prior to year 8 and that stems only are removed after that. No timber products are removed following fire, but trees are cut and left on-site, until stands are older than 8 years. Prior to age 8, losses from fire include removal of 50% understory

[10] Sixty ground control points (GCP), collected with handheld GPS receivers, were evenly distributed over the four sample areas for georeferencing. All scenes were registered to the UTM coordinate system (zone 17N), NAD 83 datum (to correspond with parcel data), and field checked for accuracy. The 30 September 1997 Landsat TM scene was chosen as the base image for the initial rectification because it exhibited good contrast between vegetation and roads (GCPs were usually taken at road intersections) and had no cloud cover. Total RMS error for the image-to-GPS-point geometric correction (first-order polynomial with nearest neighbor resampling) was 9.7 m. Subsequent images were registered to the base scene using image-to-image rectification. The average RMS error was 8.0 m for all the images with a maximum value of 8.9 m and a minimum of 6.8 m. Georeferenced images were then radiometrically calibrated using the CIPEC method [*Green et al.*, 2005] and subset into the four sample areas.

[11] We estimated landscape-wide carbon storage with a look-up table of biomass/carbon content for different land cover classes (Table 1), on the basis of measurements made by previous and current projects in the study area as well as data from other studies of similar ecosystems [e.g., *Gholz and Fisher*, 1982; *Clark et al.*, 1999; *Clark et al.*, 2004; *Thornton et al.*, 2002]. The NEE portion of the look-up table was derived from eddy covariance data that were collected from the Florida Slash Pine AmeriFlux sites [*Clark et al.*, 1999, 2004; *Powell*, 2002; *Powell et al.*, 2005]. For years during which eddy covariance data were not collected, NEE was estimated from changes in ecosystem carbon content over time on the basis of a chronosequence study by *Gholz and Fisher* [1982]. Losses associated with cutting and fire removal were estimated using observations made at a number of sites in the study area over the past two decades (H. Gholz and G. Starr, unpublished data, 2005). We assumed that there are no timber removals prior to year 8, because it is not cost effective. Prior to age 8, losses from fire were estimated to include 50% of the carbon in the understory, 25% that in branch biomass, and 50% that in litter. For older stands, the total also included harvested stems [*Clark et al.*, 2004].

[12] We assigned an age to every parcel of forestland determined to be under plantation in every image by determining the timing of clear-cutting. We did not use a more traditional land cover classification because of difficulties distinguishing important age classes of the pine plantations. A spectrally derived land cover classification distinguished four different growth stages of plantation forests (clear-cut, 1–3 years, 4–7 years, and $\geq$ 8 years), two different categories of forested wetlands or hardwoods

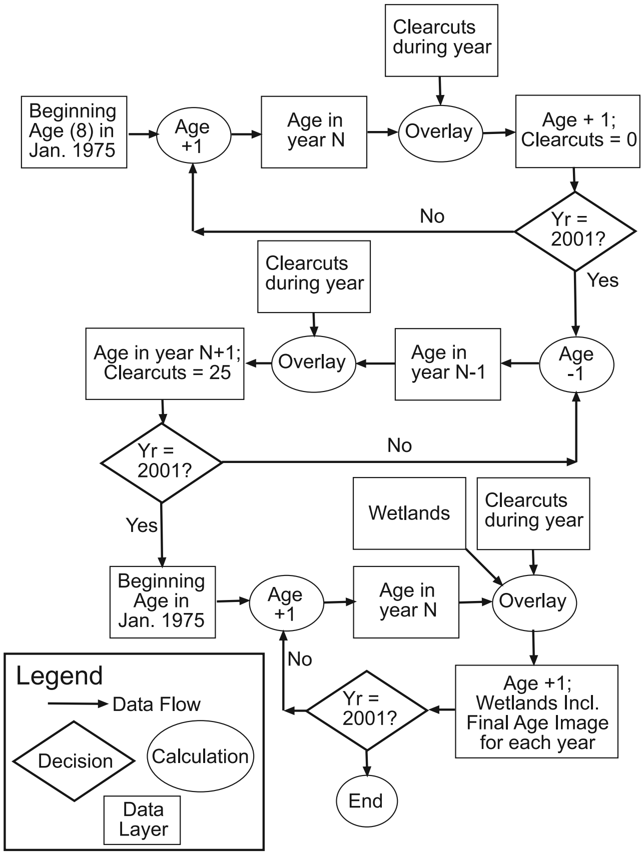

Figure 2. Flowchart of procedure for age calculation.

(wetland mixed and cypress), and three categories of non-forested land ("urban," agriculture, and open water), but was not sufficiently accurate for landscape-wide carbon exchange estimates [*Genc*, 2003]. While 1–3 and 8+ year classes were determined with 88 and 86% accuracies, respectively, the 4–7 year class was defined with only 22% accuracy. It is very difficult to use spectral methods to distinguish 3-yr-old from 4-yr-old plantations, and likewise 7-yr-old from 8-yr-old plantations. The 4–7 year growth period is when carbon exchange switches from negative (the forest is a net source of carbon to the atmosphere) to positive, making this an important time of forest growth for estimating carbon exchange [*Gholz and Fisher*, 1982; *Thornton et al.*, 2002]. Furthermore, the pattern of carbon uptake changes significantly between 8 and 20 years of age, when the forest matures and biomass accumulation rates level off [*Gholz and Fisher*, 1982].

[13] Clearing and regrowth is easily detected in successive Landsat images by visual analysis. The Normalized Difference Vegetation Index (NDVI) was calculated for all images of the study areas and NDVI differences were calculated by subtracting the more recent from the previous scene. Areas clear-cut between two Landsat scenes have large, positive NDVI differences. We then used a region-growing tool in the image analysis software to identify the full extent of the cleared areas on the NDVI difference image. NDVI difference thresholds for determining whether pixels were in the cut area were manipulated until the entire area was determined by inspection to be completely covered. Typically, the visually detectable NDVI difference threshold varied from 0.10 to 0.20, depending on the nature of surrounding vegetation. The region-growing tool uses the contrast between adjacent pixels to determine whether a pixel belongs in the "region." Older surrounding forests contrasted more than clear-cut areas or younger forests adjacent to them. NDVI of pixels with younger forests was variable and required inspection and comparisons with true color imagery to determine whether the grown "region" had included true cleared areas or young forest vegetation.

[14] When all cleared areas in each study area were identified and a data layer of the cleared areas created, the new data layer was overlaid on the 5-4-3 RGB composite image. Whether forest clearing was a result of clear-cutting, fire, or mining was determined visually by "swiping" or "flickering" back and forth from the image to the cleared area layer and separating areas with different causes of clearing into different data layers. Clear-cut areas appeared homogeneous and had regular boundaries while fires resulted in areas that appeared mottled and had irregular boundaries, so the two clearance mechanisms could be easily distinguished. Mined areas were obvious as all vegetation was cleared, borders were very regular, the vegetation never returned, and the resultant mines were often filled with water.

[15] Every 30-m plantation pixel in the sample area images was then assigned an age (time since clearing) by a multistep procedure (Figure 2). We used winter scenes (December, January, or February) to determine clearing time and considered any clearance between two successive scenes. To begin the aging method, an age of 8 years was assigned uniformly to the landscape for the first year of the calculation (1975) because we had no indication of stand age at the beginning and some age had to be assigned to start. Although 8 years seems arbitrary, it is the stand age when the canopy of a slash pine plantation usually closes completely. The exact age of older plantations cannot be determined using standard RS methods. Then, one year was added to the initial age data layer to create the 1976 age data layer, except that all the clearings were assigned an age of 0. Then another year was added to the age and the next year's clearings were reset to 0, and so on for every year until 2001. At this point, areas that were not cut from 1975 to 2001 were assigned to a "natural regeneration" forest category and excluded from the next two steps. Assuming a maximum rotation length of 25 years [*Fox et al.*, 2004], all plantation forests should have been cut during the study period.

[16] As the forest age was set to the arbitrary value at the beginning of the procedure, none of the forests except for those actually 8 years old were accurately aged until they were cleared. The final 2001 forest age calculation has accurate ages across the landscape because it is the 26th year and even the youngest plantations in 1975 would have been cut, but earlier calculations are inaccurate for forests younger than 8 years in 1975. To rectify this problem, we back calculated beginning with 2001, by subtracting one year and assigning an age of 25 years to forests that were cut in a given year. This calculation was repeated back to 1975. However, the changing rotation length and other management practices that might have caused clearing

before 25 years may have invalidated the assumption of a 25-year cutting cycle. So, we ran the aging with a clearing calculation forward again beginning with the back-calculated 1975 ages instead of a uniform, arbitrary age. Some small inaccuracies no doubt occurred in the 1975 back-calculated ages, caused by the difference between actual age of cutting and the assumed 25 years. Most of these errors are removed with the second forward calculation. Another source of error is that some cleared stands are not replanted in the first year after cutting, resulting in a true age one or perhaps two years younger than the calculated age.

[17] We do not believe that the extent of delayed planting is very great in this region because the economics of plantation forestry create strong incentives for rapid replanting [*Cubbage et al.*, 1991; *Fox et al.*, 2004]. Net ecosystem exchange of carbon (NEE), obtained using standard eddy covariance techniques, and carbon distributions were available for the full range of plantation ages [*Clark et al.*, 1999, 2004], so a value for both NEE and carbon loss to cutting or fire could be assigned to each plantation pixel in each image. Nonplantation forests (i.e., any forests not cleared during the study period) were classified as either "natural regrowth pine" or wetland forests. Wetland forests were determined using 1994–1996 GAP Analysis land cover maps from *Pearlstine et al.* [1998]. We used existing data [*Clark et al.*, 1999; *Powell*, 2002; *Powell et al.*, 2005] to assign NEE and carbon contents to natural regrowth forests and wetlands.

[18] Wetlands and natural regrowth forests were then added back into the landscape data layers and NEE calculated for each pixel using Table 1. We assigned NEE values of 0 to agricultural (crops and pasture), extractive (mining) land covers, urban lands, including roads and permanently open areas such as airports and school grounds, and nonproductive forest (sandhill). We argue that these systems are near steady state year-to-year, or if not, that they change in total carbon contents only in very small amounts. Agricultural systems in this landscape are mainly pastures under low-intensity grazing by cattle; we therefore assume that inputs to maintain grass biomass are balanced by export of animals and animal products. Conversion of forest to mining was directly counted as a carbon loss from the forested state to land with 0 NEE. This clearly does not account for any loss in soils associated with the mining, for which we have no data. With the exception of strip mining in Hamilton County, the four study areas underwent no or very small amounts of conversion from forest land to agriculture or urban land covers. Carbon losses from fires or clear-cuts were calculated by zonal analysis using the cleared areas as zones and the carbon loss values in Table 1. We calculated landscape-wide carbon loss to cutting by substituting the value in the "losses to cutting" column of Table 1 to each pixel, depending on the age of the forest. The loss to fire was calculated similarly. The total net carbon exchange (NEE in growing forest + losses to cutting in clear-cut areas + losses to fire in burned areas) was then summed across the entire landscape. NEE values were then substituted on the basis of plantation age to create a carbon exchange map. The clearing data sets (cutting, fire, and mining) were then overlain on the forest age calculations to determine loss of carbon for that particular age of forest that was cut or burned.

[19] The final results were two maps for each 15x15 km sample area for each year, one of the distributions of NEE, and one of the distributions of carbon loss to cutting and fires. The total NEE and carbon losses for each year were then summed across each map and summarized in the figures.

4. Results

[20] Our approaches to quantifying land cover explicitly accounted for the dynamics over time in some categories, but not in others. For example, we defined "pine plantation" as any forest area that was clear-cut at least once at any time during the study period. Since we started clear-cut detection in 1972, stands as old as 28 years were accounted for; this exceeds the longest normal rotation lengths in this area over this period [*Brown*, 1999; *Fox et al.*, 2004]. Clear-cuts were the most easily, accurately identified land features in the images. Many parcels were cut twice and some even three times. On the other hand, "wetlands" were defined only once using results from the Florida GAP Study [*Pearlstine et al.*, 1998], which was based on analysis of satellite remote sensing and airborne videography data from 1993–1994. Federal and Florida wetland protection laws were first enacted in 1972, making it unlikely that wetlands were converted to another land cover during our study period. Occasionally we observed that small areas of wetlands were cut or burned, but never more than 7–8 ha. When they were cut we then regarded them as either forest (if trees regrew) or agriculture/urban (if no trees regrew). Conversion of forest to mining was important over only a few years in one of the four areas (Hamilton County). Coverage of urban, agriculture, open water, and nonproductive forest (sandhill) was held constant at 2000 levels; the latter two categories were very small in total cover and simply did not change.

[21] We expected more detectable urbanization, especially in Clay County, which borders the Jacksonville development area. However, even there, not much conversion took place in our study area through 2000, although a few short roads were established. Most of the urban conversion that did occur began from agriculture, not forest. We had no data on urban carbon budgets and it is difficult to predict the directions in which development would take them. Starting from agriculture, where the carbon balance was assumed to be neutral, carbon storage could occur as trees are planted for landscape planting, a common path for development in this region. However, excavations for housing and paved roads could remove carbon stored in soils and eliminate potential sinks in ground vegetation. Even ignoring the fossil fuel component, the resulting carbon balance over time was not possible to predict given our current state of knowledge and measurements. The assumption of continued neutrality seems reasonable in this context, although this is clearly an area requiring greater attention in future research.

[22] The proportion of the region classified as forest was consistent across the four sample areas, averaging 67% (Table 2). Pine plantations covered from 27–42% of the sample areas and averaged 33% cover, the same as that of the combined wetland and other upland forests.

[23] The four sample areas contrasted greatly in their patterns of carbon gain, loss and net exchange over time

Table 2. Areas of Each General Land Cover Type in the Four Study Parcels[a]

	Omitted/Assigned 0	Plantation Forest	Wetlands	Mixed Pine/Mixed Forest	Total Forested	Total Area, ha
Alachua	8924 (39)	7251 (32)	3492 (15)	3045 (13)	13,788 (61)	22,712
Clay	7328 (34)	5846 (27)	2948 (14)	5443 (25)	14,237 (66)	21,572
Hamilton	5962 (28)	6769 (31)	3676 (17)	5113 (24)	15,558 (72)	21,521
Union	7196 (32)	9599 (42)	4420 (19)	1546 (7)	15,563 (68)	22,760
Average, %	(33 ± 5)	(33 ± 6)	(16 ± 2)	(17 ± 9)	(67 ± 5)	

[a]Figures in parentheses are percentages of the total area for each parcel. The column "Omitted/Assigned 0" lists the area of each study site that was not considered in the calculations or was assigned 0 NEE (urban and agricultural areas).

(Figures 3a–3d). However, it is clear that the interannual variation in landscape net carbon exchange is dominated by annual changes in the extent of plantation harvesting (Figures 4 and 5). This observation alone highlights two important aspects of forest carbon dynamics.

[24] First, we argue that, on the basis of previous work, neither the normal seasonal droughts nor the severe droughts of 1986–1987 and 1999–2001 had significant effects on annual net carbon exchange. There was no explicit consideration of drought effects on stand level C flux in our procedure, but both previous research [*Teskey et al.*, 1994; *Albaugh et al.*, 1998; *Cropper and Gholz*, 1993] and direct evidence from the tower flux measurements through the latter drought suggest that water deficit has

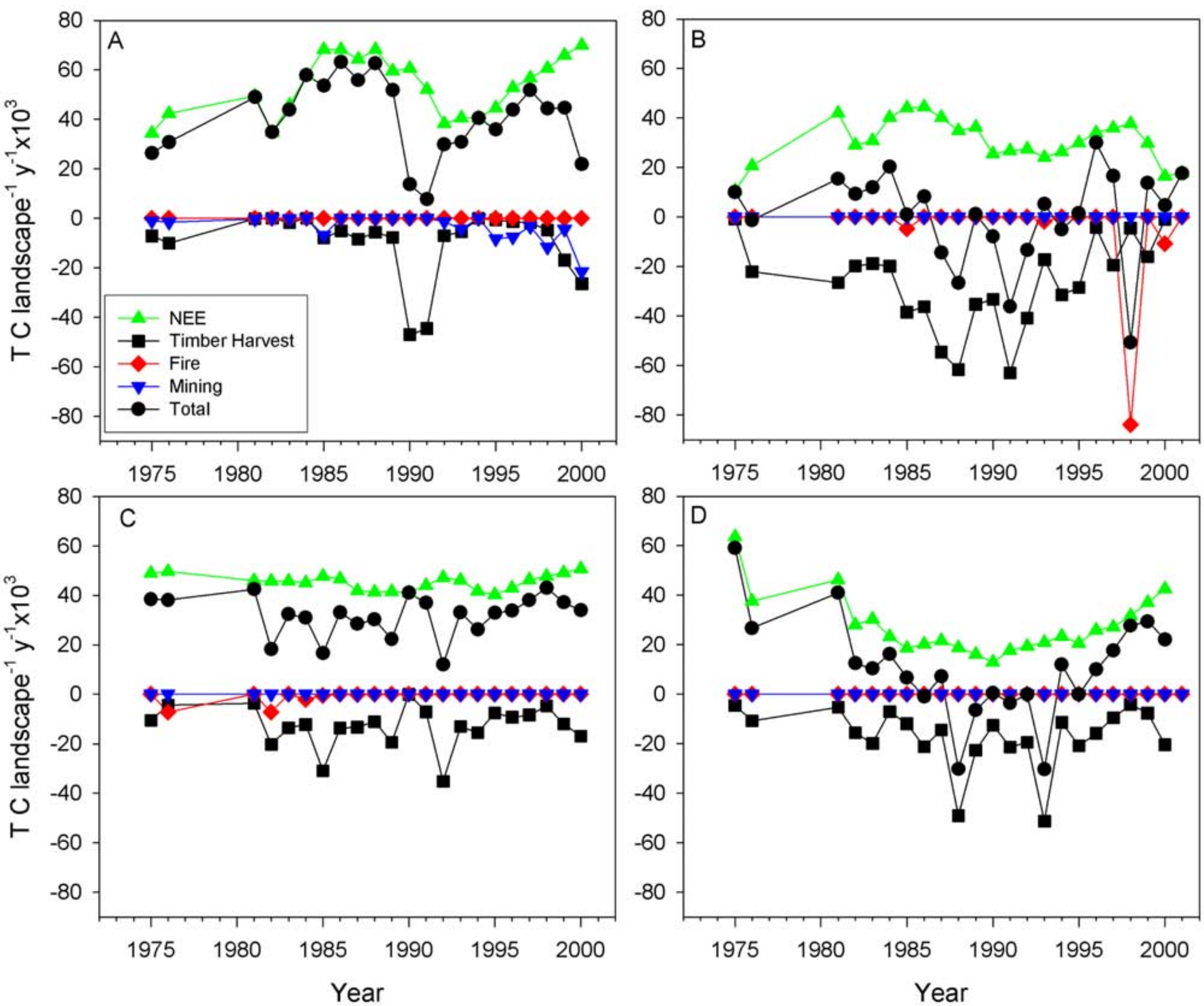

Figure 3. Carbon exchange time series for four sample areas, the study areas in (a) Hamilton, (b) Alachua, (c) Clay, and (d) Union counties. NEE is the net ecosystem exchange determined for each pixel as a function of forest age from the look-up table (Table 1), summed across the landscape. Cutting, fire, and mining are the C losses from each pixel, again as a function of forest age, from the look-up table, and summed across the landscape. Total C exchange is landscape NEE minus landscape losses.

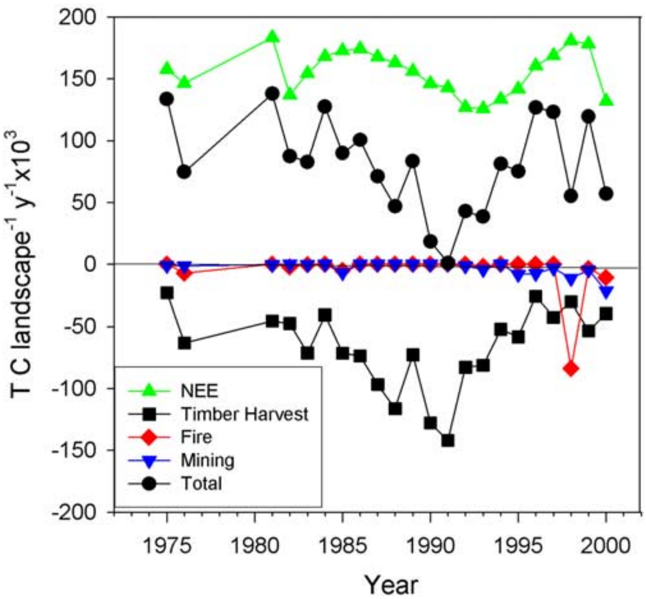

Figure 4. C exchange time series for total of all four sample areas. The legend entries are defined as in Figure 3.

minimal effects on net C exchange in these systems. The tower measurements revealed that drought reduced the magnitude of gross carbon gain but also reduced C losses due to ecosystem respiration. These offsetting effects resulted in a relatively small (12%) decline in NEE as a result of the prolonged drought. In fact, there is little indication that changes in environmental conditions, *per se*, affected our carbon exchange results, although landscape level NDVI did appear to be reduced during these two prolonged droughts (data not shown) and measurements during the more recent drought showed that LAI of the AmeriFlux natural pine site was also reduced [*Powell et al.*, 2005]. Our results are qualitatively similar to those from a recent study of the extreme 2003 drought in Europe. *Ciais et al.* [2005] showed proportional declines in both gross carbon gain and total ecosystem respiration, despite elevated temperatures that occurred during the drought. Our relatively smaller quantitative reduction in the NEE of the Florida pine ecosystems under drought may be related to lower soil carbon contents and relatively low decomposition rates [*Gholz et al.*, 2000].

[25] The primary influence on the variability of annual landscape carbon exchange was land use; recent clear-cuts are highly negative while rapidly growing plantations are highly positive. *Clark et al.* [2004] noted this pattern from flux tower observations, commenting that the range in NEE on the same Florida pine site from year to year over a cutting event, or between two adjacent sites where one was recently cut and the other supports a growing stand, are greater than the full range of NEE values reported in the extant literature. This highlights the need to identify carefully the time it takes for newly planted land to switch from a source to a sink for carbon. There may be important indirect effects of precipitation on the spatial patterns of forest harvesting, in that forest managers plan for fluctuations in soil water conditions by holding some mature stands on dryer sites in reserve and moving harvesting operations to them during wet periods. However, this compensation was sufficiently local to not show up in our analysis as a net effect on the cutting rates at the sample area scale.

[26] Second, the two obvious outliers in Figure 5 are interpretable on other than environmental grounds. The first (less deviating) reflects a one-time clearing of large amounts of forest in the Hamilton County site for mining in 2000. The second (more deviating) marks the extensive "Waldo Fire" in the Alachua County site in 1998. Both wildfires and mining are disturbances characteristic of the region, both have a strong human element of control (most wildfires are caused by arson), and both can result in large ecosystem carbon releases. The impacts of mining are permanent, while ecosystem recovery after fires is rapid because of a rapid rate of natural successional processes [*Gholz and Fisher*, 1982] and rapid reforestation. In any case, both fire and mining are highly local in space and time, which limits their impacts at the landscape level as seen when the four study areas are combined (Figure 4).

[27] The annual sums of the processes we included in this analysis indicate that there is an average annual, region-wide sink for carbon on the order of +1 t ha^{-1} yr^{-1}, but with large uncertainty and spatial and interannual variability. Averages across the sample areas ranged from near zero (Alachua County) to as much as 1.9 t C ha^{-1} yr^{-1} (Hamilton County), with annual values ranging from −2.2 (Alachua County in 1998, mostly a consequence of the Waldo Fire) to +2.9 (Hamilton County in 1986 and 88). Little timber harvesting took place in Hamilton County within those two years. The age analysis indicates that large areas of mature pine stands were cut in both years for unknown reasons (most of this tract is in large blocks owned by one family, according to county tax assessor's records). Aside from the recent mining and these two heavy harvesting years, the Hamilton County study area was a

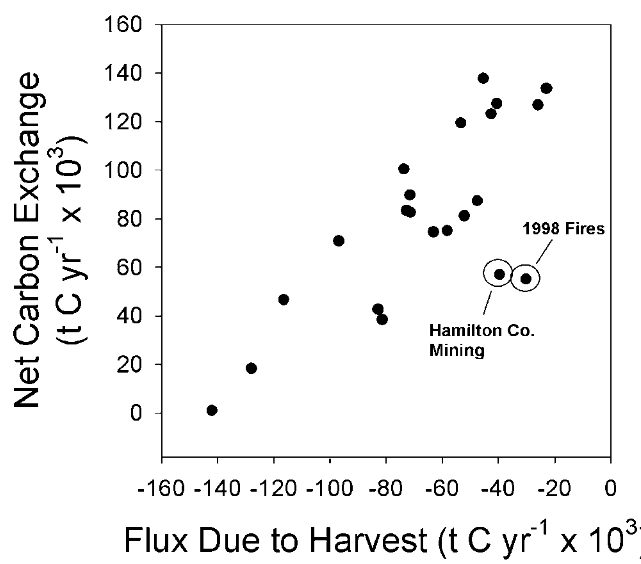

Figure 5. Relationship between net carbon exchange and timber harvest removals for the four 15 × 15 km sample areas aggregated (i.e., 900 km^2) for the duration of the study (1975–2001). Each point is the total carbon exchange (net C exchange from Figure 4) plotted as a function of C loss due to cutting (cutting losses on Figure 4).

Table 3. Sensitivity Analysis of Consequences of Potential Age Assignment Error[a]

Timing of Error	Area in Error Calculation Assuming All Pixels Are Aged Accurately	10% of Pixels 1, 2, or 3 years Too Old	20% of Pixels 1, 2, or 3 years Too Old	30% of Pixels 1, 2, or 3 years Too Old	40% of Pixels 1, 2, or 3 years Too Old	50% of Pixels 1, 2, or 3 years Too Old
1-year NEE % error	0	0.70%	1.30%	2.00%	2.70%	3.50%
1-yr difference t C ha^{-1}	0	−0.02	−0.03	−0.05	−0.07	−0.09
2-year NEE % error	0	−1.41%	4.82%	−2.21%	−4.21%	4.99%
2-year difference t C ha^{-1}	0	0.04	−0.13	0.06	0.11	−0.13
3-year NEE % error	0	1.59%	−0.86%	−4.14%	0.52%	15.75%
3-year difference t C ha^{-1}	0	−0.04	0.02	0.11	−0.01	−0.41

[a]The analysis uses the age map of January 1997 and assumes that all age assignment errors are too old, e.g., a recently clear-cut plantation forest assigned to 3 years may actually be 2 years old because of a delay in replanting. Younger errors are not possible. NEE is modeled for landscape in errors at 10, 20, 30, 40, and 50% of the pixels being incorrectly assigned dates of 1, 2, and 3 years too old. The errors do not systematically increase with percent wrong pixels or number of years wrong because of the specific age distribution of the 1997 landscape. Other years would have slightly different values.

consistent carbon sink with consistently low levels of cutting otherwise. Temporal variability was more muted and almost flat in Clay County. Overall Clay County averaged about +2 t C ha^{-1} yr^{-1} (Figures 4 and 5). Harvesting was heaviest over time, but also highly variable, in Alachua County, with Union and Clay counties intermediate (Figures 3a–3d). We conclude that fluctuations in landscape carbon exchange are most broadly related to plantation harvesting, with important but restricted effects of local disturbances, and a small contribution from environmental fluctuations.

5. Discussion and Conclusions

[28] Our study aggregated C dynamics in similar-aged stands across space and did not create a physical model of C exchange based on familiar parameters such as Leaf Area Index (LAI) or fraction of absorbed photosynthetically active radiation (f_{APAR}). None of the physical models are very successful currently [e.g., *Goetz and Prince*, 1998] except perhaps for grasslands [*Wylie et al.*, 2002]. Instead, our approach depends on extensive measurements in forests across a full management cycle. Aggregation according to forest age does allow the incorporation of spatial heterogeneity of C dynamics and the effects of various disturbances on them that operate at different spatial and temporal scales (e.g., fire, logging).

[29] As with all C cycle estimates, these calculations are subject to several sources of uncertainty, principally errors of age designation, variability of stand level eddy covariance measurements, and stand-to-stand variability. A sensitivity analysis that examined age-related error using the 1997 age map showed that errors of assigning ages to pixels were insignificant except at the worst case extreme of 50% of the pixels incorrectly assigned an age of 3 years (Table 3). We argue that this level of age-related error is unlikely, again because of the economically driven industry practice of rapid replanting.

[30] Uncertainty of measured annual NEE is on the order of 5% for stand level eddy covariance methods [*Hollinger and Richardson*, 2005], and perhaps as large as 30% for stand-to-stand measurements (G. Starr and T. Martin, unpublished and preliminary data, 2005). A sensitivity analysis of our method, again using the January 1997 age map indicated that constant 5, 10, 20, and 30% (standard deviation) NEE uncertainties resulted in per hectare NEE errors of 0.23, 0.47, 0.93, and 1.40 t C ha^{-1} yr^{-1}, respectively. Our overall suggestion of 1 t C ha^{-1} yr^{-1} net uptake by forests across the landscape is greater than the uncertainty except at the highest levels (30%) of stand-to-stand error. Given the large number of stable nighttime hours that are estimated indirectly, an error this large in annual NEE from individual tower measurements is possible when the eddy covariance approach is used, although there is no reason that we can envision for this being systematically positive or negative. Stand-to-stand variation may be a systematic error source, given the relatively few stands for which we have made tower flux measurements, although again, we can see no reason why NEE in the stands we measured would be systematically biased.

[31] The imbalance that we estimate, +1 t C ha^{-1} yr^{-1}, may actually be correct, although it cannot currently be

validated. Fire (including its absence), is the most characteristic and dominant historic influence on forest characteristics in this region [*Myers and Ewel*, 1990]. Low-intensity wildfires maintained the more open natural stands. Low-intensity fires were used at the beginning of our study period as a site preparation tool to reduce logging debris during plantation establishment, and fire losses were included as part of the chronosequence carbon budgeting of *Gholz and Fisher* [1982]. Low-intensity prescribed fires were also used into the 1980s in midaged plantations to reduce understory and litter accumulations to reduce wildfire risk and improve access for forest management activities (e.g., ground fertilizer applications). Active fire suppression and fire avoidance policies were followed for the past two decades, mainly because of the potentially large economic and legal consequences of concentrated smoke plumes and escape of the fire itself [*Fox et al.*, 2004]. Large-scale fires (either low in intensity but large in space, or intense but local, as the Waldo Fire we included in this study) in this landscape are now rare. In fact, our study period is the first in which wildfire has not been widespread and prescribed fires only rarely used as a forest management tool. So although we cannot verify it with current data, we conclude that forests of all types in this landscape are, in fact, accumulating carbon.

[32] The apparent landscape carbon sink can perhaps be best understood in the context of an ecosystem carbon balance. A net sink of 1 t C ha^{-1} yr^{-1} is the difference between net primary production and heterotrophic respiration, where NPP itself is the difference between gross primary production (essentially the sum of carbon fixed by photosynthesis over the entire area) and autotrophic respiration. In carbon balance terms, NPP can be broken down into carbon that accumulates in living plants (mainly as wood in tree stems in our plantation-dominated case, but also as coarse woody roots, branches and bark), finer components of litterfall (mainly leaves) deposited onto the soil surface, and root turnover into the soil detrital pools. The wood accumulated in stems of the plantations is eventually removed from the study areas during harvests, and for our purposes represents a once-per-rotation net regional loss of carbon. The other organic accumulations, both above and belowground and including accumulated litter and understory biomass as well as woody debris, remain on-site after harvest. The aboveground material is either plowed into piles or rows or left spread out and physically reduced in size mechanically to facilitate machine planting of seedlings. *Gholz and Fisher* [1982] proposed that the sum of this residual carbon on a unit of pine plantation land remained relatively constant over time through successive rotations, while the woody vegetation carbon increased with growth and was lost at harvest. This means that each ecosystem was thought to be in steady state with respect to carbon over a long time period (i.e., multiple rotations). However, they assumed that logging debris was scattered, mechanically broken up ("chopped"), and broadcast burned, which resulted in a large carbon release that they indirectly assessed by mass balance. Our present analysis indicates that the region is accumulating carbon, despite continual wood removals, occasional wildfires, and mining conversion, albeit in a highly variable spatial pattern. Accumulation occurred in both the plantation portion of the landscape, as well as in the portion covered by unmanaged forests, although about two-thirds of the total appears to have been on plantation land, even though this accounts for only one-third of the total landscape area (Table 2). Apportioning two-thirds of +1 t C ha^{-1} $yr^{-1}$to the plantation area roughly translates to 1300 kg of organic matter ha^{-1} yr^{-1}. This represents an equivalent to about 25% of average annual needle fall, or 10% of average aboveground net primary production [*Gholz et al.*, 2000; *Powell*, 2002; *Powell et al.*, 2005]. Belowground NPP (or fine root production) is roughly equal in magnitude to aboveground NPP in these stands [*Gholz et al.*, 1986]. Collectively this means that the carbon accumulation rate we observed is equivalent to about 5% of aboveground NPP. It is questionable that this is an empirically measurable rate of organic matter accumulation, given current available technologies and methods. We are currently conducting further measurements to expand the range of the tower flux measurements, as well as the associated ecological measurements, to more stands in the region to provide stand-to-stand replicates and reduce the uncertainties.

[33] Our analytical approach may be applicable at scales larger than our landscape. *Barnett and Sheffield* [2004] show the stand age distribution of slash pine in 2000 for the entire growing area from North Carolina through east Texas. Assigning the mean NEE values from Table 1 to Barnett and Sheffield's stand age data and summing over all age groups, results in an estimate of 17.7×10^6 t C fixed by southeastern U.S. slash pine ecosystems in 2000. This amount can be verified from their statement that "The current (2000) net annual growth of slash pine growing stock totals 871 million cubic feet..." This growth occurred over the 4.2 million ha listed as the area of timberland classified as a slash pine forest type [*Barnett and Sheffield*, 2004; Table 1]. If we assume that each cubic foot of slash pine weighs 19 kg [*Wang et al.*, 1982] and that wood is 50% C, then there were a total of 8.3 million t C, or 1.98 t C ha^{-1} in this growth, which is about twice that estimated using our methods for the total landscape, but similar to that of one of the study areas (Hamilton). *Barnett and Sheffield* [2004] do not include any land cover type other than slash pine plantations. Adding wetlands, agriculture, urban, and natural regrowth forests with much lower rates of accumulation would decrease the difference between the two results. These two estimates then agree that the southeastern U.S. coastal plain is a net sink for carbon on the order of 1–2 t C ha^{-1} yr^{-1}.

[34] The scale of our study is intermediate between plot based or eddy covariance tower studies on the one hand and most of the studies that attempt regional models of NPP with satellite data. Nearly all of the previous work at the regional level uses estimates of LAI and the fraction of photosynthetically active radiation (f_{PAR}) derived from NDVI calculations from data collected by instruments, such as the Advanced Very High Resolution Radiometer (AVHRR – 1.1 km), MODIS (250, 500, and 1000 m), and SPOT-4 VEGETATION (1.14 km), with 1-km spatial resolution or larger [*Veroustraete et al.*, 1996; *Vourlitis et al.*, 2003; *Xiao et al.*, 2004]. The calculations, which are spatially explicit, are then input to ecosystem models to estimate spatial variation in C exchange. We do not model the ecophysiology of the system, but use a simplified look-

up table approach and a much finer spatial resolution of 30-m cells. Spatial resolution of all but one of the other studies [*Sun et al.*, 2004] is coarser than the disturbance scale of our landscape (clear-cuts are typically 20–100 ha). Most of the clear-cuts and fires are smaller than single pixels of the coarser instruments, so will not be detected with the instruments. Thus other approaches must be used for estimating C exchange and its heterogeneity in both space and time at the landscape level.

[35] We suggest that the north Florida study sites, although presented as a case study, may be representative of the carbon dynamics of managed forests of the U.S. southeastern coastal plain. Soils, climate, vegetation, and human activities differ somewhat across the coastal plain, but not as much as one might expect. With regards to vegetation communities of the coastal plain, *Christensen* [1999, page 436] states: "…community types identified with specific habitats are quite similar, even though some sites were over 2000 km from each other." We argue that studies done in a specific habitat on a specific vegetation types in the southeastern coastal plain will have broad applicability.

[36] Our approach should be applicable to other regions if C measurements are made in a full range of vegetation types or characteristics that can also be determined and mapped over time using satellite remote sensing. The values for NEE, biomass losses to disturbances, and their uncertainties will be different, but the method should still provide insights into landscape-wide patterns of C dynamics.

[37] Carbon budgets can be constructed at scales from subcellular through global, but measurements at any one scale do not necessarily enable predictions of budgets at other scales. Extrapolating from small-scale measurements within any particular ecosystem to the landscape, or region, ignores important spatial and temporal environmental heterogeneities, and new methods must be devised for estimates at each level. Endogenous factors, such as variations in soil characteristics or topography, and exogenous disturbances are not considered when small-scale measurements are scaled up to larger areas, but they lead to regional variations in carbon distribution and dynamics over space and time. These factors must be considered when reporting data in publications and in public forums, and different but appropriate methods must be developed for carbon balance estimates at each scale level considered.

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