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Contemporary Carbon Dynamics in Terrestrial Ecosystems in the Southeastern Plains of the United States

SHUGUANG LIU*
SAIC
EROS Data Center
Sioux Falls, South Dakota 57198, USA

THOMAS R. LOVELAND
RACHEL M. KURTZ
US Geological Survey
EROS Data Center
Sioux Falls, South Dakota 57198, USA

ABSTRACT / Quantifying carbon dynamics over large areas is frequently hindered by the lack of consistent, high-quality, spatially explicit land use and land cover change databases and appropriate modeling techniques. In this paper, we present a generic approach to address some of these challenges. Land cover change information in the Southeastern Plains ecoregion was derived from Landsat data acquired in 1973, 1980, 1986, 1992, and 2000 within 11 randomly located 20-km × 20-km sample blocks. Carbon dynamics within each of the sample blocks was simulated using the General Ensemble Biogeochemical Modeling System (GEMS), capable of assimilating the variances and covariance of major input variables into simulations using an ensemble approach. Results indicate that urban and forest areas have been increasing, whereas agricultural land has been decreasing since 1973. Forest clear-cutting activity has intensified, more than doubling from 1973 to 2000. The Southeastern Plains has been acting as a carbon sink since 1973, with an average rate of 0.89 Mg C/ha/yr. Biomass, soil organic carbon (SOC), and harvested materials account for 56%, 34%, and 10% of the sink, respectively. However, the sink has declined continuously during the same period owing to forest aging in the northern part of the ecoregion and increased forest clear-cutting activities in the south. The relative contributions to the sink from SOC and harvested materials have increased, implying that these components deserve more study in the future. The methods developed here can be used to quantify the impacts of human management activities on the carbon cycle at landscape to global scales.

Atmospheric carbon balance studies have suggested that about 2–4 Pg (Pg = 10^{15} g) of carbon must have been absorbed by terrestrial ecosystems each year in the 1990s, although the locations and the mechanisms of this sink are not clear (IPCC 2001, Schimel and others 2001). A recent study on the status of the world's humid tropical forest cover using satellite remote sensing imagery revealed that the net rate of loss of forest cover was 23% lower than the generally accepted rate between 1990 and 1997, implying that the terrestrial carbon sink might be smaller than previously inferred (Achard and others 2002). Therefore, developing consistent, spatially explicit, and accurate land-use and land-cover change (LUCC) databases over large areas is one of the key issues to improve our understanding of the dynamics of the carbon cycle at regional to global scales.

KEY WORDS: Carbon cycle; Source and sink; Land cover change; Modeling; Uncertainty; Scaling

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*Author to whom correspondence should be addressed, *email:* sliu@usgs.gov

The US terrestrial ecosystems might contribute a significant portion to the global carbon sink (Ciais and others 2000, Hurtt and others 2002, Houghton and others 1999, Pacala and others 2001). However, uncertainty remains as to the magnitude, the locations, and the underlying mechanisms of the US sink. The net carbon flux from the atmosphere to land in the conterminous United States was estimated to be 0.37–0.71 Pg C/yr during the 1980s (Pacala and others 2001). This estimate included all the possible C changes in vegetation and soils (including woody encroachment), wood products, carbon burial in depositional environments (reservoirs, alluvium, and colluvium), food and wood imports and exports, and exports to oceans by rivers. The range or uncertainty of the carbon sink estimates was probably underestimated in Pacala and others (2001) because the uncertainties of some components might be larger than the values provided. For example, 0.11–0.15 Pg C/yr was given as the range of the sink due to forest tree biomass increment, which was much higher than the estimate of Houghton and others of 0.06 Pg C/yr (Houghton and others 1999). Similarly, the range (0–0.04 Pg C/yr) given to cropland soils was too small to contain the estimate of Houghton and others of 0.14 Pg C/yr. In addition, the magnitude of sink due to woody encroachment was in fact highly uncertain, and yet its range was given as 0.12–0.13 Pg C/yr. More studies are needed to reduce the uncertainty of the estimates of the contemporary carbon sink in the Unites States and to understand the relative contributions from different sectors and processes.

Although many biogeochemical models have been developed to simulate C cycles at various spatial and temporal scales in recent years, few of them have the capability of dynamically assimilating LUCC information into the simulation processes over large areas (Chen and others 2000, Melillo and others 1995, Potter and others 1993, McGuire and others 1997, Pan and others 1998, Schimel and others 1991), partly owing to the lack of detailed LUCC information. Previous process-based modeling studies over large areas have been done mostly under potential vegetation conditions, with relatively static land cover and land use information, or the ecosystem models were applied to very coarse spatial resolution, which could result in biases (Kimball and others 1999, Pierce and Running 1995, Turner and others 2000, Jenkins and others 2001). It is therefore a high priority to develop methods for assimilating various detailed spatially explicit information into process-based biogeochemical modeling processes over large areas.

The ultimate goal of this study was to quantify the contemporary temporal and spatial patterns of C sources and sinks in the conterminous United States. In this paper, we present the results for the US Southeastern Plains ecoregion (Omernik 1987). The paper covers (1) the overall methods for quantifying spatial and temporal patterns of the LUCC and carbon dynamics in the conterminous United States, (2) the trends of LUCC in the Southeastern Plains from 1973 to 2000, and (3) the contemporary trends and uncertainties of carbon sources and sinks in the Southeastern Plains.

Methods

Introduction to the Contemporary US Carbon Trends Project

The contemporary US Carbon Trends project deserves a brief introduction because the work in the Southeastern Plains reported in this paper represents an integral part of the project. It differentiates itself from previous studies (Pan and others 1998, Hurtt and others 2002, Potter and others 2001, McGuire and others 2001) in three major ways: (1) it is based on the detailed investigation of carbon dynamics within sampling blocks rather than the conventional wall-to-wall approach, (2) land cover and land use change was detected using remote sensing data at a high temporal sampling frequency and high spatial resolution, and (3) carbon dynamics within each of the sampling blocks was simulated using the General Ensemble Biogeochemical Modeling System (GEMS), capable of assimilating the variances and covariance of major input variables into simulations using an ensemble approach.

Sampling strategy and land cover/use change Wall-to-wall simulations of terrestrial carbon dynamics in the conterminous United States are possible but are frequently prohibited by the lack of consistent high-resolution LUCC databases. In the US Carbon Trends project, a sampling rather than a wall-to-wall approach was used. This sampling framework was inherited from the Land Cover Trends project (Loveland and others 2002).

LUCC is one of the predominant forces that change the properties of land surfaces and the biogeochemical interactions with the atmosphere. However, it has been a major challenge to detect and quantify the dynamic nature of LUCC on the Earth's surface over large areas. Initiating a program to develop periodic, detailed (i.e., including land cover and land use types and transitions), wall-to-wall mapping of LUCC for the United States at a temporal interval appropriate for determining types, distributions, rates, agents, and consequences of change is cost prohibitive. The Land Cover Trends adopted a more feasible and cost-effective sampling strategy to study the contemporary LUCC in the conterminous United States (Loveland and others 2002). Sampling blocks were randomly selected for each of the 84 Omernik level III ecoregions (Omernik 1987) to identify changes with 1% precision at an 85% confidence level. Change detection was based on five dates of Landsat Multi-Spectral Scanner and Thematic Mapper data (i.e., 1973, 1980, 1986, 1992, and 2000) analyzed at a common cell size of 60 m.

Model simulation of carbon dynamics Carbon dynamics in vegetation and soils is simulated using the GEMS at the spatial scale of 60 m in each of the 20-km × 20-km sampled blocks. GEMS is a modeling system that was developed for a better integration of well-established ecosystem biogeochemical models with various spatial databases for the simulations of the biogeochemical cycles over large areas (Figure 1). It uses a Monte-Carlo-based ensemble approach to incorporate the variability (as measured by variances and covariance) of state and driving variables of the underlying biogeochemical

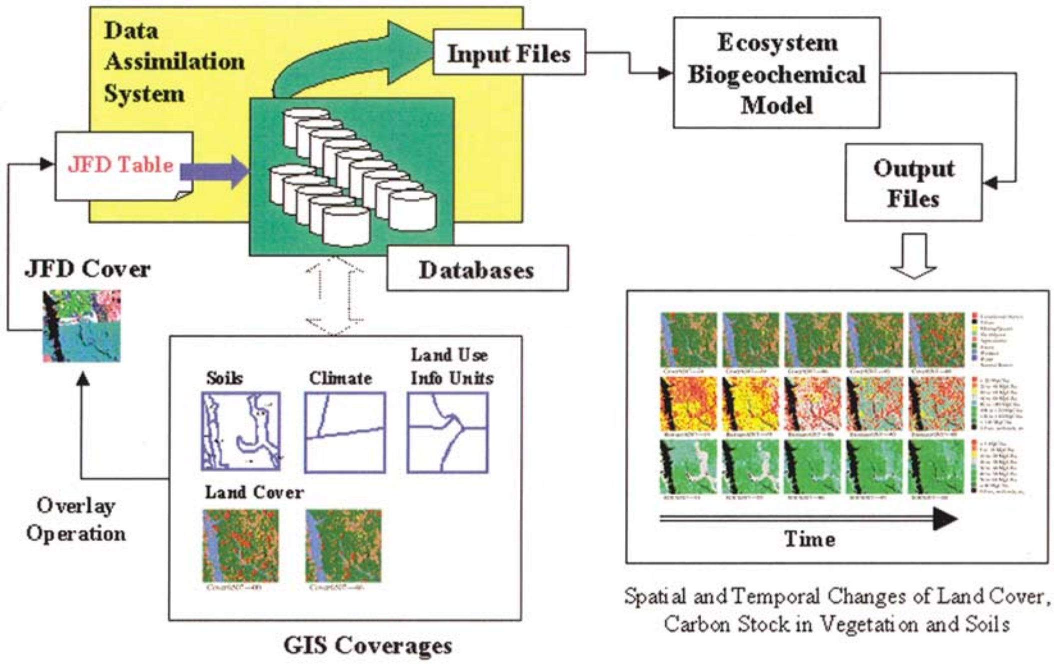

Figure 1. Diagram of the General Ensemble Biogeochemical Model System (GEMS) and major steps in the application of the GEMS over large areas.

models into simulations. Consequently, GEMS can not only simulate the spatial and temporal trends of carbon dynamics such as CO_2 exchange between the terrestrial biosphere and the atmosphere, but can also provide uncertainty estimates of the predicted variables in time and space. A prototype of GEMS was successfully used for scaling nitrous oxide emissions from sites to the entire Atlantic Zone in Costa Rica (Reiners and others 2002).

GEMS consists of three major components: one or multiple encapsulated ecosystem biogeochemical models, a data assimilation system (DAS), and an input/output processor (IOP). The CENTURY model was selected as the underlying ecosystem biogeochemical model in GEMS for this study because it has solid modules for simulating carbon dynamics at the ecosystem level and it has been applied to various ecosystems, including crops, pastures, forests, and savannas worldwide (Parton and others 1987, 1994, Schimel and others 1991, 1994, Pan and others 1998). CENTURY simulates C, N, P, and S cycles in various ecosystems with the capability of modeling the impacts of management practices, including land cover change, fertilization, and cultivation, and natural disturbances, such as fire and hurricane (Parton and others 1994, Ojima and others 1994, Del Grosso and others 2002, Liu and others 1999). However, the application of CENTURY in limited forests (Ryan and others 1996) and our preliminary application in the Southeastern Plains indicated that its forest growth modules need to be improved. Because most forests in the region are young and growing, it is important for a model to reasonably capture the dynamics of forest growth to provide accurate estimates on the dynamics of carbon sources and sinks. Details on the modifications of the model are given in the appendix.

Most information in spatial databases is aggregated to the map-unit level, making the direct injection of such information into the modeling processes problematic and potentially biased (Kimball and others 1999, Reiners and others 2002). Consequently, a DAS is usually needed for incorporating field-scale spatial heterogeneities of state and driving variables into simulations. A DAS generally consists of two major interdependent parts: (1) data search and retrieval algorithms and (2) data processing mechanisms. The first part searches for and retrieves relevant information from various databases according to the keys provided by a joint frequency distribution (JFD) table (Reiners and others 2002). The data processing mechanisms downscale the aggregated information at the map-unit level to the field scale, using a Monte Carlo approach. More details of the data assimilation processes are given in the appendix.

Once the data are assimilated, they are injected into the modeling processes through the IOP, which updates the default input files with the assimilated data. Values of selected output variables are also written by the IOP to a set of output files after each model execution.

The encapsulated biogeochemical model within GEMS is deployed in space with the consideration of the covariance of major input variables to estimate the regional state or flux of a certain biogeochemical variable, as follows:

$$E(p) = \int E[p(\mathbf{X})]f(\mathbf{X})\,dX \quad (1)$$

where E is the operator of expectation, p is the encapsulated biogeochemical model, X is a vector of model variables, and f is the joint probability density function (pdf) of X. It is assumed in this equation that the spatial biogeochemical interactions (i.e., the interactions among plots) are insignificant and that the joint pdf of X is generated at the spatial scale of the model, namely, the specific spatial scale from which the model was developed and successfully tested.

It is usually impossible to analytically integrate equation 1 because of model complexity. In practice, the major input variables are often partitioned in space into discrete, "homogeneous" regions using geographic information system (GIS) techniques so that equation 1 can be modified as a summation over a JFD table of the major input variables X:

$$E(p) = \sum_{i=1}^{n} E[p(\mathbf{X}_i)]\,\mathrm{F}(\mathbf{X}_i) \quad (2)$$

where n is the number of strata or JFD cases as defined by the GIS overlays of the major input variables, and F is the frequency of cells or the total area of stratum i as defined by $\mathbf{X}_i$.

Any difference between the model scale and data scale is subject to introducing biases because biogeochemical models are nonlinear (Kimball and others 1999, Pierce and Running 1995, Reiners and others 2002). An ensemble approach is used in GEMS to assimilate the fine-scale heterogeneities in the databases in order to reduce the potential biases. The ensemble approach emphasizes the assimilation of the fine-scale data by performing multiple model simulations for each coarse-scale stratum as defined by the JFD table:

$$E[p(X_i)] = \frac{1}{w}\sum_{j=1}^{w} (\mathbf{X}_{ij}) \quad (3)$$

where w is the number of fine-scale model runs for stratum i, and X_{ij} is the vector of model input values at the fine scale generated using a Monte Carlo approach. Input values for each model run are sampled from their corresponding potential value domains usually described by their statistical information, such as moments and distribution types.

The Southeastern Plains

The Southeastern Plains ecoregion is located in the Southeastern part of the United States and has a total area of 33.5 million ha extending from Mississippi to northern Florida to northern Virginia (Figure 2). Millions of acres of forests in the Southeast are maturing on land that was previously agricultural (Birdsey and Heath 1995). This area is ideal for intensively managed pine plantations because the warm wet climate and favorable soil conditions result in rapid tree regeneration.

A total of 11 sampling blocks 20 km × 20 km were randomly selected within the ecoregion for land cover change detection and carbon dynamics analysis. On the basis of the 11 sample blocks, the Southeastern Plains landscape in 1973 consisted of four major land cover types: forests (53%), agriculture (24%), wetlands (10%), and urban and residential areas (9%). The land cover composition in the samples matched well with that in the entire ecoregion, although the sampling blocks only covered 1.29% of the ecoregion.

Deriving carbon stocks and NPP from FIA databases The estimates derived from the Forest Service's Forest Inventory and Analysis (FIA) database (http://www.fia.fs.fed.us/) were used in this study for model validation [e.g., net primary productivity (NPP) and C stock] and model parameterization (e.g., forest age distribution, mortality, and selective cutting). The stock and annual increment of aboveground live biomass carbon density, natural mortality, selective and clear-cut harvesting were derived from all the FIA field plots of 15 inventories (spanning from 1970 to 1995) in the southeastern United States. The following methods were used to derive NPP and total biomass C from the FIA-based aboveground estimates.

NPP NPP for a given forest inventory plot was estimated using the common assumption that root production equals fine litterfall (Raich and Nadelhoffer 1989, Jenkins and others 2001) as follows:

$$NPP = 2L + C_g \quad (4)$$

where L is annual fine litterfall, and C_g is the annual increment of aboveground woody components, including stems and branches. According to the above equation, the average NPP of forests at the state level can be estimated by:

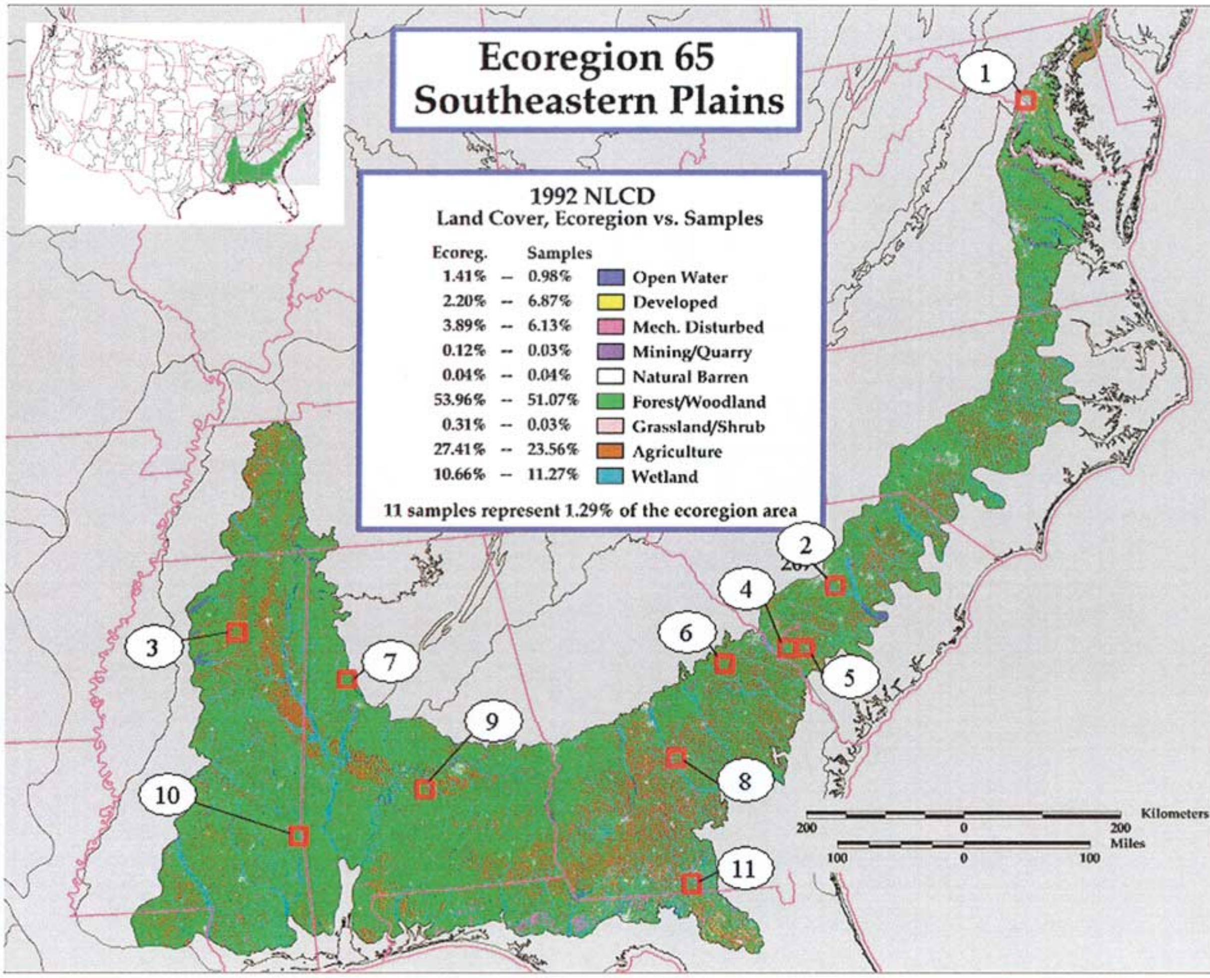

Figure 2. The land cover types within the Southeastern Plains and the locations of the 11 sampling blocks used in this study. The 11 samples represent 1.29% of the ecoregion area. The insert shows a general agreement between the land cover compositions in the entire ecoregion and that in the 11 sampling blocks. Land cover data for the entire region were based on the USGS 1992 National Land Cover Data.

$$\overline{NPP} = \bar{2}L + C^{-}_{g} \tag{5}$$

where $\overline{NPP}$, L^{-}, and C^{-}_{g} are statewide mean *NPP, L,* and C_g, respectively. The values of C^{-}_{g} for the states pertinent to the Southeastern Plains ecoregion were estimated from the FIA database. The average annual fine litter was estimated to be 2 Mg C/ha/yr in this region on the basis of Jenkins and others (2001) and Meldahl and others (1998).

Carbon stock in vegetation Belowground biomass C stock in live roots was estimated using the regression equation for temperate forests (Cairns and others 1997). Estimated belowground biomass C stock was then added to the aboveground estimate to produce an estimate of total biomass C stock.

The fate of carbon in harvested wood The annual amount of harvested wood C (HWC) can be tracked with the CENTURY model, but not its fate. In this study, a simple offline first-order decay function was applied to track the fate of the HWC:

$$K_{i+1} = (1 - \beta)\, H_{i+1} - \frac{1}{\alpha} C_i \tag{6}$$

$$C_{i+1} = C_i + K_{i+1}$$

where C_i and C_{i+1} are the HWC stock at year i and year $i + 1$ respectively, β is the fraction of the immediate HWC weight loss in the year of harvest due to wood processing, α is the turnover time (in years) of the remaining HWC that survived the immediate loss, H_{i+1} is the amount harvested in year $i + 1$, and K_{i+1} is the annual change of HWC stock.

Harvested wood can be used for various purposes and the decomposition rates of end products vary greatly (Pingoud and others 2001, Cote and others 2002, Young and others 2000). In this study, it was assumed that $\beta = 0.5$, because wood processing in mills generally results in 50%–55% of immediate loss of HWC weight owing to the extraction of lignin and organics (Cote and others 2002). The turnover time of HWC varies from a few years to hundreds of years. The half-time for carbon in each end use (i.e., the time after which half of the carbon placed in use is no longer in use) can vary from 1 year for some paper products to 100 years for houses (Skog and Nicholson 1998), suggesting that the corresponding turnover time varies from 2 to 150 years. Winjum and others (1998) used a turnover time between 25 and 100 years for different wood commodities for the temperate region. If, when taken out of use, products are disposed of in a landfill, they will stay there indefinitely with almost no decay (Micales and Skog 1997). Because tracking the end uses of harvested wood is out of the scope of this study, we used an average turnover time of 50 years for the remaining HWC after the immediate loss, which might end in usable products or landfills. On the basis of the temporal change of HWC stock at the national scale (Skog and Nicholson 1998), it was assumed that the initial HWC stock in 1973 was 20% below the average from 1900 to 2000. Sensitivity analysis was performed to investigate the impacts of turnover time and initial HWC stock on carbon sources and sinks over the entire Southeastern Plains.

Supporting databases and data assimilation mechanisms Initial soil characteristics within these sampling blocks were based on the US State Soil Geographic Data Base (STATSGO) (USDA 1994). The use of 1994 STATSGO soil data for model initialization will lead uncertainty, but we believe that this uncertainty was small. STATSGO data were based on a compilation of the soil survey data at the county level and the time of the survey varied among counties and states. Nevertheless, a majority of the published county-level data (prior to the 1994 STATSGO database) falls within the 1970s and 1980s in the southeastern states (http://www.statlab.iastate.edu/soils/soildiv/sslists/sslisthome.html). Given about 5–10 years to a life cycle of a typical soil survey (from the collection of field samples to laboratory analysis and publication), the majority of the data in the 1994 STATSGO might represent the general soil conditions in 1970s, corresponding to the start time of this study.

In GEMS, a soil component was randomly picked for a specific stochastic simulation from all components within a soil map unit according to their shares in area. The larger the area a component covers, the more likely it will be selected as the basis for a specific model simulation. Once the component was determined, soil characteristics, such as drainage class, soil texture, high and low values of soil organic matter content (converted to SOC using a factor of 0.5), soil bulk density, and water-holding capacity, were retrieved from the corresponding soil component and layer attribute databases.

To run the encapsulated biogeochemical model for any given individual plot, we first had to create a land cover and land use change file (i.e., the event schedule file in CENTURY). This file specifies the time period for the simulation, the type and timing of any LUCC change, as well as the types and timing of management practices, such as cultivation and fertilization. In GEMS, the schedule file was created on the fly before each model simulation by means of the DAS. Cropping practices, including shares of various crops and rotation probabilities, were derived from the National Resources Inventory (NRI) database, developed by the Natural Resources Conservation Service, US Department of Agriculture (http://www.nrcs.usda.gov/technical/NRI/).

Total atmospheric nitrogen deposition from wet and dry sources was from the National Atmospheric Deposition Program (http://nadp.sws.uiuc.edu/). Climate data (i.e., long-term mean monthly precipitation, maximum and minimum temperatures) were from the Vegetation/Ecosystem Modeling and Analysis Project (VEMAP) (Kittel and others 1995).

Modeling strategy and joint frequency distribution (JFD)- The impacts of CO_2 fertilization, climate change and interannual variability, and soil erosion and deposition were not simulated in this study. Wetlands and waterbodies were excluded because the underlying biogeochemical model CENTURY is an upland ecosystem model. Although management activities, such as clearcutting events, were found existing in wetlands, they were very limited in space.

A JFD grid for each of the 11 sampled blocks was generated from overlaying the following GIS grids at a common cell size of 60 m: land cover grids in 1973, 1980, 1986, 1992, and 2000; climate map units from the VEMAP, soil map units from STATSGO, county Federal Information Processing Standard (FIPS) code, and nitrogen deposition map unit (Figure 1). Each 20-km $\times$ 20-km sampling block resulted in 111,111 cells at 60-m $\times$ 60-m resolution. The numbers of strata contained in the JFD tables of the 11 sampling blocks varied from 166 to 681, much smaller than the total number of cells in the grid. Five ensemble stochastic model simulations were performed for each JFD stratum to sample the heterogeneity and uncertainty of the data that define the stratum.

Table 1. Annual rates of three most common land cover conversions during four time periods in Southeastern Plains since 1973[a]

1973 to 1980		1980 to 1986		1986 to 1992		1992 to 2000	
Type	Rate (%)	Type	Rate (%)	Type	Rate (%)	Type	Rate (%)
F2C	1.24	F2C	1.52	F2C	2.10	F2C	2.53
C2F	1.07	C2F	1.26	C2F	1.59	C2F	1.97
F2A	0.18	A2F	0.21	A2F	0.61	A2F	0.34

[a]Rates are expressed as the percentage of the forest area.
F2C: forest to clear-cut; C2F: clear-cut to forest; F2A: Forest to agriculture; A2F: Agriculture to forest.

Table 2. Percentage shares of various land cover types in the Southeastern Plains in 1973, 1980, 1986, 1992, and 2000

Category	1973	1980	1986	1992	2000
Water	1.0	1.0	1.1	1.0	1.1
Developed	8.4	8.6	8.9	9.1	9.8
Disturbed	2.2	2.4	2.9	3.8	4.8
Mining	0.1	0.1	0.1	0.1	0.1
Forest	53.4	52.5	52.0	52.7	52.5
Grass/Shrub	0.0	0.0	0.0	0.1	0.0
Agriculture	24.8	25.2	24.8	22.9	21.7
Wetland	10.2	10.3	10.3	10.3	10.0

Results and Discussion

Land Cover and Land Cover Change

Table 1 lists the three most common land cover conversions during the four time periods. Forests that were clear-cut and then reforested were the most common land cover changes. The forest area that was affected by forest clear-cut activities annually doubled from 1.24% in 1973 to 2.53% in 2000. The reforestation rate also increased from 1.07% to 1.97% during the same period. The reforestation rate during one specific time period was comparable to the precedent clear-cut rate, indicating that the forests cleared in the previous time interval were regenerated in the next, rather than being a net conversion of forest to other land covers. Conversions of agricultural land to forest ranked as the third most frequent change during three of the four time periods. However, the total area that was under forest cover remained relatively stable (Table 2). Because of the cyclic nature of timber harvesting, the total land area under tree cover has fluctuated over time between a low of 52.0% in 1986 and a high of 53.4% in 1973. Given that most of the disturbed landscape was due to forest harvesting, the total forest land (i.e., cleared plus forested) would be 55.6%, 54.9%, 54.9%, 56.5%, and 57.3% of the land area in 1973, 1980, 1986, 1992, and 2000, respectively, indicating a trend of increasing forest land use in the last decade. At the same time, cropland decreased from 25.2% in 1980 to 21.7% in 2000, and urban lands increased from 8.4% to 9.8% between 1973 and 2000.

In previous regional to global carbon studies, land cover change information was frequently based on national or regional census/survey data, such as the USDA NRI for agricultural land (Eve and others 2002, Heath and others 2002, Houghton and others 1999) or the USDA Forest Service FIA data for forests (Heath and others 2002). These data might provide inconsistent area estimates of land cover and land use change because they may be incomplete (e.g., NRI only covers private land) or are different samples and the uncertainties might be large (Heath and others 2002). For example, building a house in a small clearing that is sampled by the FIA system would result in a large area being transferred into urban use in the FIA database, even though only a very small area of forest was actually affected (Heath and others 2002). According to Heath and others (2002), the conversion from forest to agriculture was 0.07% of the forest area each year from 1977 to 1986, while our estimate was 0.18% from 1973 to 1980. Similarly, the conversion from agriculture to forest was estimated to be 0.12%–0.14% per year using the FIA data, and our estimate was 0.21%–0.61% per year. Although the Southeastern Plains ecoregion only accounted for 56% of the entire Southeastern United States as reported in Heath and others (2002), the

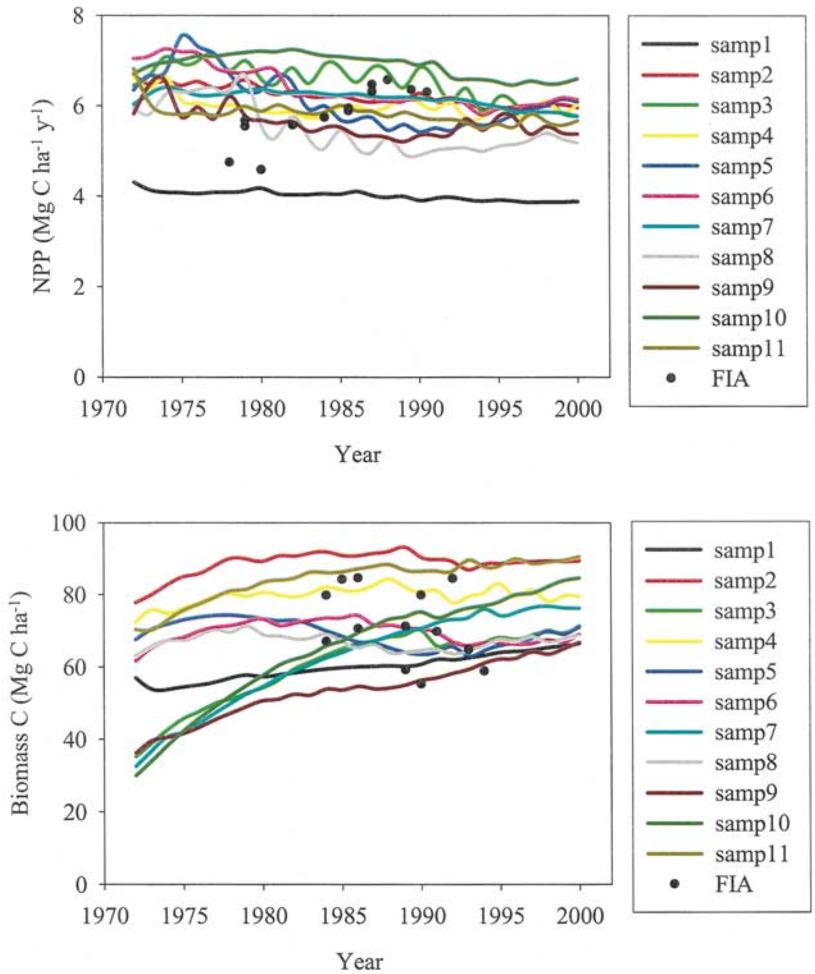

Figure 3. Model-simulated temporal changes of net primary production (NPP) and biomass C density within the 11 sampling blocks and the comparison with statewide mean values derived from the FIA database.

comparisons might indicate that land cover conversion rates were more dynamic than the FIA database suggests.

Model Validation Against Data Derived from FIA Database

Figure 3 shows the temporal changes of simulated forest NPP and C stock density within the 11 sample blocks and the comparison with the statewide mean values derived from the FIA database. It can be seen that model simulations were in general agreement with the FIA estimates. The smaller spatial variability of the FIA estimates could be explained by the fact that they were statewide averages while simulated values were means for the sampling blocks, and the spatial variability of the mean values are expected to "contract" as the spatial extent increases.

Model Simulated Forest Carbon Change

Biomass On the basis of model simulations, biomass C was relatively stable on nonforest lands, while forest biomass C has been increasing at a rate of 0.83 ± 0.15 Mg C/ha/yr on average since 1973. However, the rate has been decreasing, primarily owing to forest aging in the north and intensification of forest clear-cutting in the south. The rates during the 1970s, 1980s, and 1990s were 1.90 ± 0.24, 0.59 ± 0.13, and 0.38 ± 0.19 Mg C/ha/yr, respectively. Our estimate for the 1980s was comparable with the national average of 0.45 to 0.61 Mg C/ha/yr estimated for the same period (Pacala and others 2001, Birdsey and Heath 1995, Birdsey 1992), and higher than the 0.24 Mg C/ha/yr estimated by Houghton and others (1999). Considering that the forest biomass C increment in the Southern United States was about 30% lower than the national average (Birdsey 1992), the FIA-based estimate (Pacala and others 2001, Birdsey 1992) could be adjusted to 0.33 Mg C/ha/yr in the Southern United States. It implies that the biomass C sink in the Southeastern Plains was stronger than the average in the South.

Soil organic carbon Soil organic carbon (SOC) in forests in the Southeastern Plains ecoregion on average

Table 3. Decadal changes of carbon sources and sinks and relative contributions from biomass, soil organic carbon (SOC), and harvested wood carbon (HWC)[a]

	Time Period 1973-1979	1980-1989	1990-2000	1973-2000
Absolute sink (Mg C/ha/yr)				
Biomass	1.12 (0.17)	0.39 (0.08)	0.19 (0.10)	0.50 (0.09)
SOC	0.42 (0.04)	0.28 (0.02)	0.24 (0.03)	0.30 (0.02)
HWC	0.01 (0.04)	0.10 (0.02)	0.14 (0.04)	0.09 (0.02)
Total	1.54 (0.17)	0.78 (0.11)	0.57 (0.10)	0.89 (0.09)
Relative share (%)				
Biomass	72	51	34	56
SOC	27	36	42	34
HWC	1	13	24	10
Total	100	100	100	100

[a]Values in parentheses are 1 standard error of the mean.

increased at a rate of 0.56 Mg C/ha/yr since 1973. This indicates that SOC in the Southeastern Plains forests is still in the stage of recovery from the SOC depletion due to the agricultural development in the 1800s and the abandonment in the first half of the 20th century (Waisanen and Bliss 2002). Our estimate of SOC increment rate was similar to the national average estimated by Birdsey and Heath (1995). However, because the rate of SOC increase in the southern forests was 30% lower than the national average (Birdsey 1992), our estimate might suggest that the sink strength may be stronger than previously estimated. Heath and others (2002) estimated that afforestation in the Southeastern United States from 1953 to 1997 added very little to the SOC, which does not conflict with our estimate. Their conclusion was related to the small fraction of forests that were afforested from 1953 to 1997, while our study dealt with all the forests in the region. Nevertheless, they used a rate ranging from 0.1 to 0.7 Mg C/ha/yr to represent the SOC increase following conversion of cropland or pasture to forest in their "bookkeeping" model.

Field observations on SOC change after reforestation and afforestation vary depending on site-specific properties, such as land use history and soil texture. Paul and others (2002) reviewed SOC change following afforestation using a global database including 204 sites. They found that the data were highly variable, with SOC either increasing or decreasing, particularly in young forests. In plantations older than 30 years, SOC content within the surface 10 cm of soil was similar to that under previous agricultural systems; yet at other sampling depths, SOC had increased by between 0.50% and 0.86% per year. They also found that the most important factors affecting SOC change were previous land use history, climate, and the type of forest established. The review by Post and Kwon (2000) also found that reforestation of abandoned cultivated land generally results in an increase of SOC, although the process may take 10 to as many as 200 years. Previous observations in the Southeastern United States were consistent with these findings. For example, Johnson and Todd (1998) reported that SOC stocks increased 0.6–1.8 Mg C/ha/yr on average from 1980 to 1995 in mixed oak forests that were converted from woodland pasture around 1942 in Tennessee. Richter and others (1999) observed a significant increase in SOC at the 0- to 7.5-cm depth and a decrease at the 35- to 60-cm depth from 1962 to 1997 in eight permanent loblolly pine plantation plots, and carbon sequestration was limited by rapid decomposition facilitated by the coarse soil texture and low-activity clay mineralogy.

Harvested wood The temporal change of the HWC stock is a running balance between the input from tree harvesting and decomposition of HWC. It decreases in years with lower harvest. Our estimate of C sequestration in the HWC per hectare of forest in the Southeastern Plains was 0.16 Mg C/ha/yr on average from 1973 to 2000 (Table 3), which was comparable with the national averages (Pacala and others 2001, Skog and Nicholson 1998). By estimating the stocks and flows of carbon from US forests to products in use, to dumps or landfills, to burning and emissions from decay, Skog and Nicholson (1998) estimated that the annual carbon sequestration rate in wood products varied from 0.021 to 0.059 Pg C from 1970 to 2000 for the entire United States (rates after 1986 were projected). Given a forest area of 247 million ha (Pacala and others 2001), these C sinks ranged from 0.09 to 0.25 Mg C/ha/yr. Carbon sequestration in wood or paper products has been increasing in the Southeastern Plains because of the increase of forest harvesting activities.

Initial HWC stock in 1973 was important in determining the magnitude of C sink or source of HWC

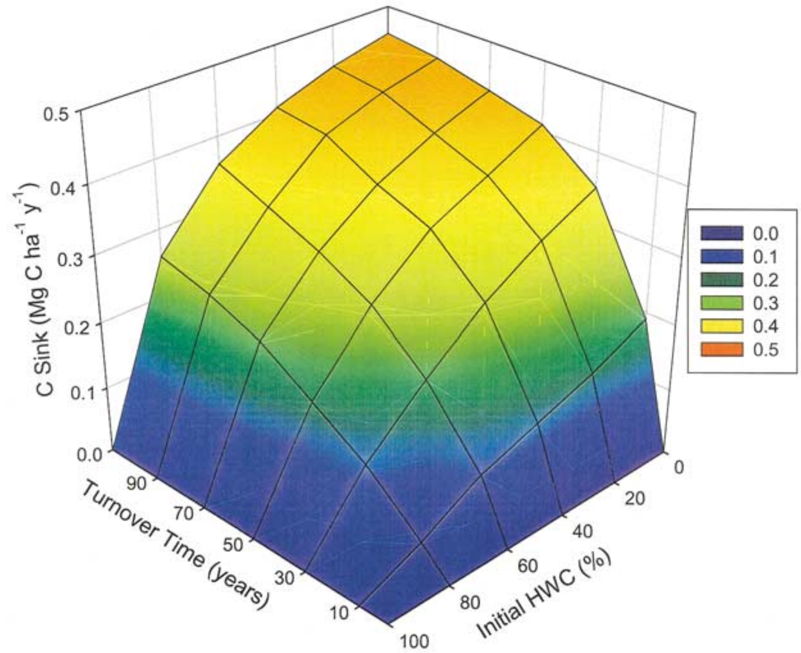

Figure 4. Model-simulated impacts of the initial HWC stock in 1973 and the turnover time of HWC on the mean annual HWC sink during the period from 1973 to 2000. The HWC stock in 1973 was expressed as a percentage of the mean annual HWC stock from 1990 to 2000.

(Figure 4). HWC would act as a C sink if its initial stock in 1973 were lower than the mean HWC stock from 1990 to 2000, and it would act as a source if it were higher. The HWC sink decreased with the increase of initial HWC stock in 1973. When the initial HWC stock was about 100% of the mean annual HWC stock of the 1990s, HWC was neither a sink nor a source regardless of its turnover rate. The facts of the increased intensity of timber harvest (clear-cut activities increased from 1.24% to 2.53% from 1973 to 2000, see Table 1) and a relatively stable share of forest cover during this period suggested that the HWC stock in 1973 was probably lower than that in the 1990s (Skog and Nicholson 1998), although the exact stock of HWC in 1973 was difficult to estimate.

As expected, the HWC sink increases with the increase of the turnover time of HWC (Figure 4). However, the rate of increase is nonlinear and decreases with the increase of turnover time. The relationship between the HWC sink strength and turnover time depends on the initial HWC stock in 1973. For a given turnover time, the sink is more pronounced if the initial HWC stock is smaller. Because the average turnover time of HWC (after the immediate loss) is usually longer than 30 years and the initial HWC in 1973 was likely higher than 50% of the annual mean in the 1990s, the accuracy of the initial HWC stock is more important than the accuracy of turnover time of HWC in determining the sink strength of HWC in the Southeastern Plains (Figure 4).

Regional Carbon Sources and Sinks

Averaging across all land cover classes shows that biomass has been accumulating C at an annual rate of 0.50 + 0.09 (1 standard error) Mg C/ha/yr on average from 1973 to 2000 (Table 3). Although biomass has been accumulating on average during the entire period and on a decadal time scale, the ecoregion lost biomass C in certain years (e.g., in the early 1990s) because biomass removal by harvesting exceeded the natural growth of the vegetation biomass in these years (Figure 5). The decadal means of biomass C increment were 1.12, 0.39, and 0.19 Mg C/ha/yr during the 1970s, 1980s, and 1990s, respectively, suggesting a dramatic reduction in sink strength over time. The biomass contribution to the total C sink decreased from 72% in 1970s to 34% in 1990s.

The average rate of SOC increase within the 11 sample blocks was 0.30 Mg C/ha/yr from 1973 to 2000 (Table 3). Soils have always been a C sink each year during this period, different from the dynamics of biomass (Figure 5). Although the SOC sink strength has been decreasing, its relative contribution to the total C sink increased from 27% in the 1970s to 42% in the 1990s.

The average HWC sink from 1973 to 2000 was 0.09 Mg C/ha/yr. HWC sink increased from 0.01 to 0.14 Mg C/ha/yr. Its share in the total C sink increased from 1% to 24% from the 1970s to the 1990s.

Summarizing the C sources and sinks of biomass, SOC, and HWC from our 11 sample blocks indicates

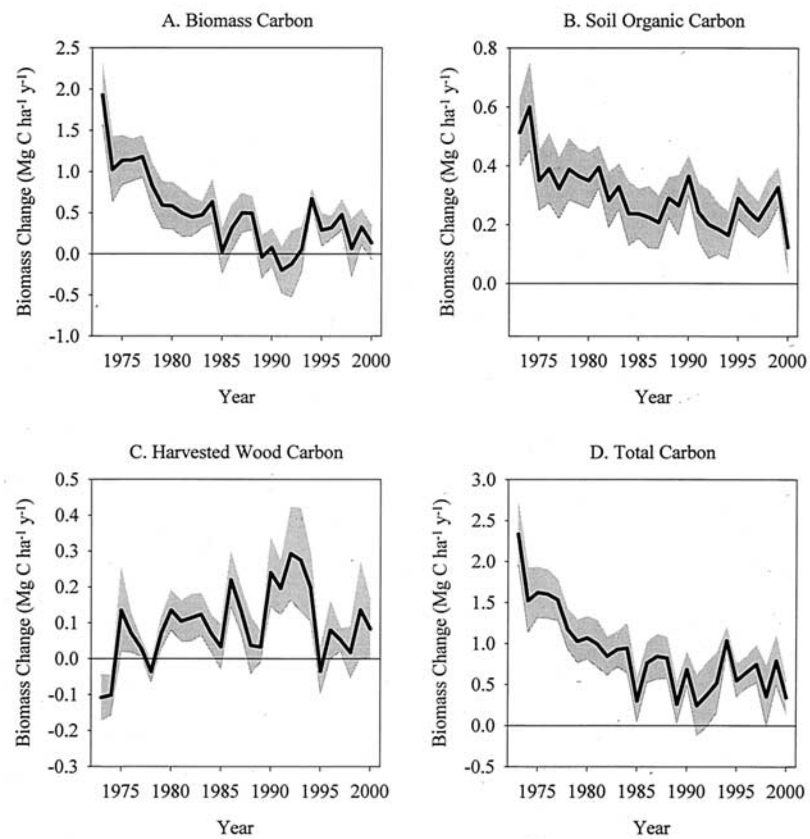

Figure 5. Net annual change rates of C stock in (A) biomass, (B) soil organic carbon, (C) harvested wood C stock, and (D) the total C stock. The shaded areas represent 1 standard error around the mean (solid line).

that the terrestrial ecosystems in the Southeastern Plains during the period from 1973 to 2000 acted as a carbon sink, on average, with a net flux of 0.89 Mg C/ha/yr. When this sink was extrapolated to the entire ecoregion, it represented a carbon sink of 0.030 Pg C/yr, or from 0.024 to 0.036 Pg C/yr with a 95% confidence level. C increments in biomass, soils, and HWC stock explained 56%, 34%, and 10% of this sink, respectively.

Summary

LUCC has a predominant role in determining the dynamics of carbon sources and sinks at spatial scales ranging from plot to the entire globe. However, mapping and quantifying LUCC over large areas have been extremely challenging and costly. The method of relying on a sampling framework rather than the conventional wall-to-wall approach for generating LUCC data is cost-effective and scale-independent. It can be used at regional to global scales.

Contemporary LUCC in the Southeastern Plains was characterized by the increases of urban and forested areas, decrease in agricultural land, and extension of forest harvesting activities. The Southeastern Plains was a C sink from 1973 to 2000 owing to forest growth on previously cultivated and abandoned land. The dynamics of C in vegetation, soils, and HWC are important components contributing to this C sink. The aging of forests in the northern part of the Southeastern Plains and the increasingly extensive use of forests for timber harvesting in the southern part changed the C sink dynamics in two ways. First, the total regional C sink has been reduced 63% from the 1970s to the 1990s. Second, the relative weight of biomass C sink decreased over time while those of SOC and HWC increased.

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Appendix 1: Modification of CENTURY

Simulation of Net Primary Production as a Function of Forest Age

In the CENTURY model, NPP is not explicitly simulated as a function of forest age. Field measurements on forest growth indicate that C or biomass accumulations in young stands, particularly plantations, are highly related to forest age (Oliver 1981, Gholz and Fisher 1982, Mencuccini and Grace 1996). Biomass accumulation for plantations that are close to rotation age and for most natural mature forests will tend toward zero, with annual aboveground net primary production becoming almost entirely composed of litterfall and whole tree mortality (Ryan and others 1997).

Mencuccini and Grace (1996) studied a chronosequence of Scots pine plantations with age ranging from 7 to 59 years old. The leaf area index (LAI) reached a maximum at 20 years old when the canopy closure began. By age 40 the overstory LAI started to decrease, and LAI at age 59 was about half of that at age 20. Aboveground net primary production (ANPP) initially increased with canopy LAI, but decreased as the stands aged. Based on Meldahl and others (1998), excluding all the locations where forest ages differed by more than 5 years among plots, we found that ANPP and needle NPP decrease exponentially from about 20 to 110 years of age. The ANNP decreased at a rate faster than the needle NPP, suggesting that aboveground woody NPP was decreasing faster than the needle NPP. This pattern is general for most even-aged forests (Ryan and others 1997). The study of Meldahl and others also indicated that needle fall decreases exponentially with age from about 20 to 110 years old. The decline of fine litterfall during forest aging has been observed in other forest chronosequences as well (Waring and Running 1998). In this study, we assume that the gradual reduction of fine litterfall during aging observed at the Meldahl et al study sites applies to the forests in U.S. Southeastern Plains.

Simulation of Temporal Changes in Allocation

The fraction of carbon allocated to each tree component (i.e., leaves, branches, stems, and fine and coarse roots) changes as a forest grows (Gholz and others 1986, Waring and Running 1998, Ryan and others 1997, Meldahl and others 1998). Initially, trees tend to allocate more photosynthates to leaves or needles and fine roots to maximize the capture of solar energy and absorbance of soil nutrients. The fractions to leaves and fine roots decrease gradually with time as more and more photosynthates are allocated to woody tissues (e.g., branches, stems, and coarse roots). After reaching a maximum value, the allocation to woody tissues decreases gradually while at the same time the fractions to leaf and fine root increase gradually. Finally, the allocation fractions reach an equilibrium condition.

The original CENTURY used only two values to represent the change of allocation fraction to a specific tree component as a forest grows from young to mature. For example, allocation fraction to stems changed suddenly from 0.10 to 0.30 in the temperate mixed forest (i.e., TMPMX in CENTURY code) when a certain mature year was reached. Because most of the forests in the Southeastern Plains are relatively young and growing, it is necessary to modify the changes of allocation coefficients with curvilinear functions to better simulate the carbon accumulation in these dynamic forests. In this study, we replaced the stepwise two-value functions of CENTURY with regression equations derived from Meldahl and others (1998) to represent the dynamic changes of allocation coefficients in the forests.

Appendix 2: Data Assimilation

Initializing Forest Age and Biomass with FIA Data

The initial structure of forest age is critical for simulating carbon dynamics at the regional scale because many characteristics of forests, including standing biomass, NPP, and allocation pattern, are closely related to forest age. In this study, we assumed that the initial forest age structure within the sample blocks follows the statewide age structure characterized by the FIA databases. If the initial land cover of a site in 1973 was forest, then its initial age is randomly assigned according to the age structure of the forests of that state. Because the time (i.e., year) when the forest inventory data were collected might not be the same as the initial time of the model simulation, an offset time (i.e., the difference between the FIA inventory year and the starting year of the simulation) was added to adjust the age estimate. Once the initial age of the forest was estimated, the corresponding standing biomass was estimated based on the relationship between age and biomass as derived from FIA data.

Initializing Soil Texture, SOC, Bulk Density and Drainage with STATSGO Data

Once a soil component was determined, information regarding drainage condition, soil layers, soil texture, water holding capacity, bulk density, and soil organic matter content (converted to SOC using a factor of 0.58) were retrieved from the STATSGO attribute databases. For the variables with high ($V1$) and low ($V2$) values, the following procedure was used to assigned a value to minimize potential bias (Reiners and others 2002, Pierce and Running 1995):

$$V = (V1 + V2)/2 + NORM*(V1 - V2)$$

where NORM is the standardized normal distribution. The above treatment assumes that the possible values of the soil characteristic follows a normal distribution with 95% of the values varying between $V2$ and $V1$. Considering the possible nonlinear impacts of texture fractions on biogeochemical cycles, a Monte Carlo approach was used to assign fractions of sand, silt, and clay in their corresponding possible ranges rather than using the mean fractions of the texture class specified in the USDA soil texture classification system.

Constructing Land Use History by Assimilating Remote Sensing, Census, and Inventory Data

It is necessary to disaggregate the agricultural land class into a combination of specific crop types for biogeochemical modeling because each type of crop has distinct biological characteristics and management practices, leading to differing impacts on carbon dynamics in vegetation and soils. Disaggregation of the agricultural land class was done stochastically based on crop composition statistics at the county level derived from USDA NRI agricultural census data (http://www.nrcs.usda.gov/technical/NRI/). The NRI database is a statistically based sample of land use and natural resource conditions and trends on US nonfederal lands. The inventory, covering about 0.8 million sample points across the country, was done once every five years.

Another task in generating LUCC sequences is to fill the gaps between consecutive land cover maps (i.e., the remotely sensed data). This was accomplished with crop rotation probabilities calculated from the NRI databases.

Our land cover change data was from the US Land Cover Trends (LCT) project (Loveland and others 2002). The LCT project was not specifically designed to capture all the forest harvesting activities. Optical reflectances at the clear-cut sites become indistinguishable from those of the surrounding mature forests 3 years after clear-cutting in the Southeastern Plains owing to fast recovery of understory and trees. The LCT data do not include selective harvesting activities. We derived the probability and intensity of selective cutting from FIA databases. The mean annual clear-cutting probability during a specific time period within a block was calculated using the LCT map by dividing the total mapped clear-cut area by 3 years. The probabilities of selective and clear-cutting were then used to stochastically schedule additional forest harvesting events that were not reflected in the LCT land cover maps. It was assumed in the model that a minimum age of 20 years was required for scheduling any harvesting activities in a forest.

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