

Effects of Forest Regrowth and Urbanization on Ecosystem Carbon Storage in a Rural–Urban Gradient in the Southeastern United States

Chi Zhang,¹ Hanqin Tian,^{1*} Shufen Pan,¹ Mingliang Liu,¹ Graeme Lockaby,¹ Erik B. Schilling,² and John Stanturf³

¹*School of Forestry and Wildlife Sciences, Auburn University, Auburn, Alabama 36849, USA;* ²*Southern Research Center, National Council for Air and Stream Improvement, Newberry, Florida 32669, USA;* ³*Southern Research Station, USDA Forest Service, Athens, Georgia 30602, USA*

ABSTRACT

Forest regrowth after cropland abandonment and urban sprawl are two counteracting processes that have influenced carbon (C) sequestration in the southeastern United States in recent decades. In this study, we examined patterns of land-use/land-cover change and their effect on ecosystem C storage in three west Georgia counties (Muscogee, Harris, and Meriwether) that form a rural–urban gradient. Using time series Landsat imagery data including MSS for 1974, TM for 1983 and 1991, and ETM for 2002, we estimate that from 1974 to 2002, urban land use in the area has increased more than 380% (that is, 184 km²). Most newly urbanized land (63%) has been converted from forestland. Conversely, cropland and pasture area has decreased by over 59% (that is, 380 km²). Most of the cropland area was converted to forest. As a result, the net change in forest area was small over the past 29 years. Based on Landsat imagery and agricultural census records, we reconstructed an annual gridded data set of land-cover change for the three counties for the period 1850 to 2002. These data sets were then used as input to the Terrestrial Ecosystem

Model (TEM) to simulate land-use effects on C fluxes and storage for the study area. Simulated results suggest that C uptake by forest regrowth (approximately 23.0 g C m⁻² y⁻¹) was slightly greater than the amount of C released due to deforestation (approximately 18.4 g C m⁻² y⁻¹), thus making the three counties a weak C sink. However, the relative importance of different deforestation processes in this area changed significantly through time. Although agricultural deforestation was generally the most important C-release process, the amount of C release attributable to urbanization has increased over time. Since 1990, urbanization has accounted for 29% of total C loss from the study area. We conclude that balancing urban development and forest protection is critically important for C management and policy making in the southeastern United States.

Key words: carbon storage; ecosystem model; deforestation; land use; urbanization; southeastern United States.

INTRODUCTION

The terrestrial carbon (C) budget in North America and the mechanisms that underlie it remain uncertain (Fan and others 1998; Bousquet and

Received 30 September 2004; accepted 24 April 2006; published online 14 August 2007

*Corresponding author; e-mail: tianhan@auburn.edu

others 2000; Wofsy and Harris 2002). Increasing forestland area and rates of production have been proposed as important mechanisms for the transfer of C between land and the atmosphere (Houghton and others 1999; Schimel and others 2000; Pacala and others 2001; Goodale and others 2002). Although forest regrowth on abandoned agricultural land enhances C sequestration, deforestation and urban development lead to terrestrial C release to the atmosphere. However, estimates of the C source and C sink induced by such land-use changes are still far from certain (Houghton 1999; Imhoff and others 2000; Milesi and others 2003; Tian and others 2003). To better quantify the roles of deforestation and forest regrowth on the regional C budget, it is essential to study landscapes where the disturbance processes are rapid and pervasive (Turner and others 1995). The objectives of this study were to characterize landscape changes along a rural–urban gradient in the southeastern United States and to determine the effects of different land uses on C fluxes and C storage in the terrestrial ecosystems of this area over the past 3 decades.

Since the middle of the 20th century, the southeastern United States has undergone a long-term transition from agricultural land use to secondary mixed forests (Hart 1980; Wear 2002). Based on young stand ages, Turner and others (1995) concluded that the southeast and south-central regions of the United States possessed the strongest biological C sinks. However, C emissions due to anthropogenic disturbances such as urbanization may essentially offset this sink. Recent analyses have indicated that trends in land development in the United States vary significantly by region (US Census Bureau, Census 2000, <http://www.census.gov/>), with the Southeast being characterized by rapid population growth and increasing urban land use. This is particularly evident in the Georgia Piedmont, where the rate of urbanization ranked among the highest in the Southeast region during the 1990s. A recent US Forest Service resource assessment (Wear 2002) indicated that urbanization will represent a primary threat to the forestland of the Southeast for the next 20 years. Furthermore, in this region, the conversion of forestland into urban land uses is counterbalanced by the conversion of cropland into forests. Thus, extensive forest regrowth on abandoned agricultural land is believed to render the Southeast a C sink. However, increased rates of urban development in the region threaten forestland and could reduce ecosystem productivity (Imhoff and others 2000; Milesi and others 2003), which would cause the Southeast to act as a C

source. Our challenge now is to accurately quantify the relative roles of forest regrowth and urbanization on the net C balance in this area.

Our approach was to use a spatially explicit process-based ecosystem model in conjunction with remotely sensed and agricultural census data to estimate annual sources and sinks of across heterogeneous landscapes along an urbanization gradient. We selected three counties in the West Georgia Piedmont (Muscogee, Harris, and Meriwether), which formed an urban–rural land-use gradient. We developed an annual gridded data set of land-use change for the three counties for the period from 1850 to 2002 by using Landsat imagery data (1974, 1983, 1991, and 2002) and agricultural census record. This data set was then used as input for the Terrestrial Ecosystem Model (TEM) to simulate land-use effects on C fluxes and C storage for the area (McGuire and others 2001; Tian and others 2003). This study of the west Georgia area served as a pilot project for the extrapolation of our analyses to the entire southeastern United States.

METHODS

Study Area

The study gradient (85°12'W/32°22'N–84°29'W/33°14'N) is located to the northeast of Columbus, the third largest city in Georgia (Figure 1). These three lower Piedmont counties have undergone significant forest regrowth since the middle of the 20th century (Hart 1980), as well as rapid population growth and related land development during the 1990s (www.census.gov). Development around Columbus is constrained by Fort Benning, a large military installation on the southeastern portion of Columbus and the Chattahoochee River to the west. Therefore, all expansion occurs in the northeastern direction. The influence of Columbus on the surrounding landscape can be understood by examining the population statistics over the past decade for the three contiguous counties. Among the three west Georgia (West GA) counties, Muscogee County, within the Columbus city limits, had the greatest population growth—41 people per km² during the study period (1974 to 2002). Conversely, Meriwether County, furthest from the city, had the lowest population growth—two people per km². Harris County, lying in between Muscogee and Meriwether counties, had a moderate population growth—nine people per km² (<http://www.census.gov/>). Thus, these three counties form an urbanization gradient that can be used to study

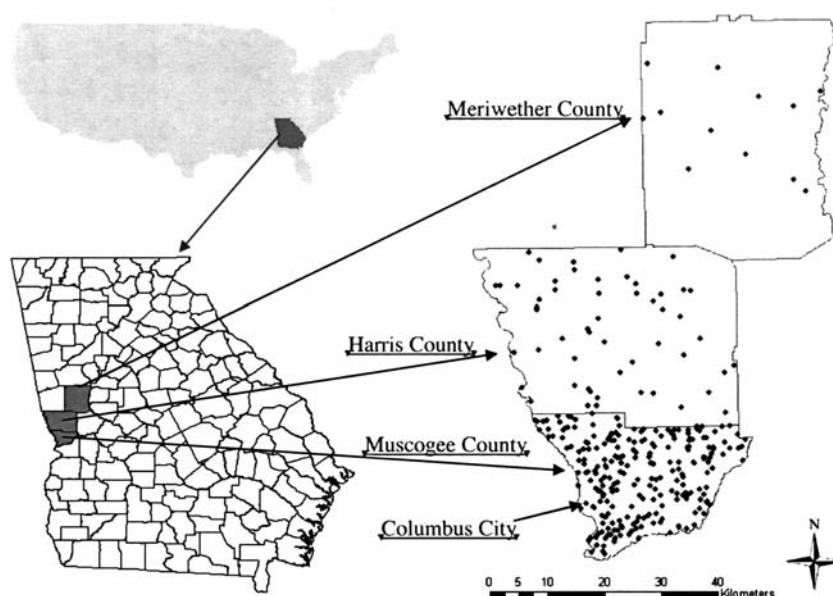


Figure 1. Location and population growth of the west Georgia (West GA) research site. From 1974 to 2002 population density of Muscogee County increased 41 people per km², population density of Harris County increased nine people per km², and population density of Meriwether County increased two people per km². The dot density in the figure is used to visualize the relative magnitudes of population growth of the three counties. The three counties, from the southwest to the northeast, form an urban–rural gradient.

the effects of land-use change on the ecosystem C balance.

The Model and Simulation Experiments

We used the TEM (Tian and others 1998, 1999, 2003; McGuire and others 2001; Felzer and others 2004) to simulate changes in C storage during three stages of disturbance; (a) conversion from natural vegetation to cultivation (cropland conversion), (b) production and harvest on cultivated land, and (c) abandonment of cultivated land (cropland abandonment). For a detailed discussion of the structure of the TEM model, refer to Tian and others (2003).

Annual net C exchange (NCE) between the terrestrial biosphere and the atmosphere can be described by the following equation:

$$\text{NCE} = \text{NPP} - R_H - E_{\text{NAD}} - E_{\text{AD}} - E_p \quad (1)$$

where NPP is net primary production, R_H is heterotrophic respiration, E_{NAD} represents emissions associated with nonanthropogenic disturbance, E_{AD} represents emissions from anthropogenic disturbances, and E_p represents the decomposition of products harvested from ecosystems for human use. We do not currently have the capability to simulate the C dynamics associated with E_{NAD} in this region; therefore, for this study, we estimated NCE by modifying Eq. (1) to:

$$\text{NCE} = \text{NPP} - R_H - E_{\text{AD}} - E_p \quad (2)$$

For regional extrapolations with TEM, we use spatially explicit data sets for vegetation, elevation, soil texture, mean monthly temperature, monthly

precipitation, and mean monthly cloudiness. The input data sets were gridded at a resolution of 1 km. In addition to the input data sets, TEM also requires that soil- and vegetation-specific parameters be assigned to a grid cell. Although many of the parameters in the model are defined from published information, some of the vegetation-specific parameters are determined by calibrating the model to the fluxes and pool sizes of intensively studied field sites. The data used to calibrate the model for different vegetation types are documented in previous studies (Tian and others 1999). To apply TEM to a transient scenario, it is first necessary to run the model to equilibrium with long-term baseline climatic data for the initial year of the simulation. Detailed documentation on the development, parameterization, and calibration of this version of TEM has also been previously published (Tian and others 1999, 2003).

To focus on the effect of land-use changes, we used a constant climatic data set—the average climate for the period from 1974 to 2002—and only allowed land cover to change over time in the simulation. We first ran the model to equilibrium with a natural vegetation map; this was derived by using a contemporary land-cover map but substituting the potential vegetation types in the area for currently cultivated and urban land. For the land-use transient run, annual historical land-use data for the period between 1850 and 2002 were used as inputs. In this paper, however, our analysis focuses on the three most recent decades (1974 to 2002), a period covered by Landsat imagery data. It is important to note, that however, the following assumptions were made for urban land uses: (a) the

intensive urban and transportation land type in the land-cover maps (see following section) are impervious surfaces (Arnold and Gibbons 1996), and (b) the impervious surfaces have zero NPP and zero vegetation cover.

Development of Gridded Annual Land-cover Data Sets

Land-cover Classification Based on Landsat Imagery. Landsat images were collected for 4 years (1974, 1983, 1991, and 2002). We used a post-classification-comparison approach to derive land-cover information from four Landsat images for the area. To improve the consistency of the classification results, all the images were georegistered to a 2003 orthophoto of the region (Lockaby and others 2005). Water bodies were identified using the supervised classification method (Jensen 1996). Next, the water patches were masked out, and the rest of the pixels were classified using ISODATA unsupervised classification methods. A total of 50 clusters were generated in the unsupervised classification. These clusters were assigned into three land-cover types: urban/transportation impervious surface, cropland/pasture, and forest. The ancillary data sets that we used included 1:24,000-scale USGS Digital Elevation Model (DEM) data (<http://www.ned.usgs.gov/>), USGS Land Use and Land Cover (LULC) data sets and the USGS National Land Cover Data set 1992 (NLCD92) data set (<http://www.edc.usgs.gov/products/land-cover.html>), as well as local transportation and hydrologic maps (<https://www.gisl.state.ga.us>). Classification accuracies were assessed using aerial photos of the study region. Accuracy of the 2002 land-cover map was assessed using a 2003 aerial orthophoto (Lockaby and others 2005). Accuracy of the 1991 land-cover map was assessed using 1993 Digital Ortho Quarter Quads (DOQQs) (<http://www.usgs.gov/>; <https://www.gisl.state.ga.us/>). Finally, the accuracy of the 1983 and 1974 land-cover maps was assessed using 17 USGS high-resolution scanned aerial photos (<http://www.edc.usgs.gov/products/aerial/hiresscan.html>), which were ortho-corrected using the 1:25,000 USGS DEM data set (<http://www.ned.usgs.gov/>) and the 1993 DOQQs orthophoto (<http://www.usgs.gov/>). Two hundreds points were randomly selected on the reference aerial photos for assessment of the accuracy of each land-cover map.

Table 1 shows the classification error matrixes and user/producer accuracies for the four land-cover maps (1974, 1983, 1991, and 2002) derived from the Landsat images. The overall accuracies of

the four land-cover maps range from 84% to 90%. Producer accuracy of forestland, the major land-cover type in the region, exceeded 90%. User accuracy of forestland exceeded 85%. In all four land-cover maps, producer accuracy of cropland/pasture was the lowest (68%–77%), mainly due to the misclassification of cropland/pasture land covers into forest type. This error of input associated with the land-use data set could result in an overestimation of soil C storage (Guo and Gifford 2002), aboveground C storage, and NPP. These effects, however, are not significant because of the slow rate of vegetation and soil C accumulation in newly abandoned croplands. For example, if a cropland pixel in the 1974 Landsat image was misclassified as forest type, forest would begin to regrow in that location some time between 1974 and 1982 in our simulation. If this pixel was correctly classified as cropland in the 1983 land-cover map (the chance of a pixel being misclassified in two consecutive images was very low), its land-cover type would be turned back into cropland in 1983. Therefore, this misclassification caused a pseudo-forest regrowth for about 1–8 years. In such a short time, the NPP, the total vegetation C, and especially the soil C accumulation rate of a newly abandoned cropland could not increase much.

The user accuracy for cropland/pasture in 1991 is only 67%. Of those reference plots that were incorrectly classified as cropland/pasture type, 80% were forest plots and 20% were urban plots. The consequences of misclassifying urban lands as cropland/pasture will not be significant because both types are disturbed ecosystems that have relatively low NPP and low vegetation C. The misclassification of forest as cropland will cause an overestimation of agricultural deforestation area, which in turn will result in overestimation of the C release due to agricultural conversion by our model simulation. Table 1 shows that of the 62 sample plots that were classified as cropland/pasture in the 1991 and 2002 land-use maps, 11 plots were actually forest type. Our analysis further shows that, of these 11 misclassified plots, nine were forest clear-cuts or newly established forest plantations. In the Landsat images, clear-cut plantations or forest stands in seedling stage were difficult to distinguish from the pasture land-use type. According to the USDA Forest Service Forest Inventory and Analysis (FIA), about 3% of the West GA forest was clear-cut or newly established forest (<http://www.fia.fs.fed.us>). As a result of this classification error, our simulation will misattribute considerable C released due to plantation rotation to negative NCE due to agricultural deforestation.

Table 1. Error Matrix of the Land-use Classification

	Land-use Types	Reference Data ^a					User Accuracy
		Water	Urban	Forest	Cropland/Pasture	Total	
1974 Classified Data Background	Water	19		3		22	86%
	Urban		7	2	1	10	70%
	Forest			105	14	119	88%
	Cropland/Pasture		1	10	38	49	78%
	Total	19	8	120	53	200	
Producer Accuracy		100%	88%	88%	72%		Over all Accuracy 84%
1983 Classified Data Background	Water	25				25	100%
	Urban		24	1	3	28	86%
	Forest		3	111	7	121	92%
	Cropland/Pasture		1	4	21	26	81%
	Total	25	28	116	31	200	
Producer Accuracy		100%	86%	96%	68%		Overall Accuracy 90%
1991 Classified Data Background	Water	14				14	100%
	Urban		20	2		22	91%
	Forest		1	127	6	134	95%
	Cropland/Pasture		2	8	20	30	67%
	Total	14	23	137	26	200	
Producer Accuracy		100%	87%	93%	77%		Overall Accuracy 90%
2002 Classified Data Background	Water	16		2		18	89%
	Urban		28	3	1	32	88%
	Forest		2	105	11	118	89%
	Cropland/Pasture		2	2	28	32	88%
	Total	16	32	112	40	200	
Producer Accuracy		100%	88%	94%	70%		Overall Accuracy 88%

^aSource of reference data for 2002: 2003 aerial photos of West GA region (Lockaby and others 2005).

Source of reference data for 1991: 1993 DOQQs (<http://www.usgs.gov/>; <http://gis1.state.ga.us/>).

Source of reference data for 1983 and 1974: nine USGS high-resolution scanned aerial photos in early 1980s and eight USGS high-resolution scanned aerial photos in the early 1970s (<http://edc.usgs.gov/products/aerial/hirescan.html>), which were ortho-corrected using 1:25,000 USGS DEM data set (<http://ned.usgs.gov/>) and 1993 DOQQs orthophoto (<http://www.usgs.gov/>).

These two kinds of disturbances, both of which involve clearcutting have similar effects on the regional ecosystem C cycle. Therefore, this classification error is not likely to affect the estimate of the total C balance in the study region.

Because the spatial resolutions of Landsat MSS (1974) and Landsat TM (1983, 1991, and 2002) differ, bias with regard to the estimates of land-use change over the study area is possible. To reduce the error of the spatial mismatch, we therefore further aggregated the simulation resolution to 1 km.

Annual Time Series of Land Cover across the Study Area 1974–2002. Land-cover maps were used to create an annually gridded data set for the period from 1974 to 2002. There are a total of 3,110 grid pixels at a resolution of 1 km in each land-use map. To generate this data set, we first aggregated

the 30-m (80-m for 1974) resolution land-use maps into 1-km resolution using majority rule (Jensen 1996) and recorded the area fraction of each land-cover type in each grid pixel. Second, for those years between two adjacent remote-sensing time periods, we constructed land-cover maps by linear interpolation so that in each year a fixed number of grid pixels will change their land-cover types. The probability of land cover change would take place in a grid pixel is positively related to the fraction of the destination land cover type in that grid pixel.

Reconstruction of Historical Land-cover Change of the Study Area 1850–1973. We reconstructed 1-km resolution historical land-use maps for each year between 1850 and 1973 in two steps. First, we estimated the cropland, urban area, and forest area in each year based on historical census data

(Waisanen and Norman 2002); then we generated land-use maps for each year based on the total area of each land-use type and the 1974 land-use map.

We estimated the historical urban area (before 1974) by assuming that urban area per capita remained constant between the 1850s and the 1970s. With this assumption, we first derived the per capita urban area based on the 1974 population census data and the 1974 land-use map. Then we estimated the total urban area in each year from 1850 to 1973 by multiplying its population size with the per capita urban area constant. For those years that have no population census data available, a linear interpolation was done based on urban areas of the nearest two census years. The database developed by Waisanen and Morman (2002) provides historical cropland area census data back to 1850. For years that have no cropland census data available, we used linear interpolation based on cropland areas of the nearest two census years. We assumed that the area of water land-use type did not change from 1850 to 1973. Therefore, the forest area could be estimated by subtracting water area, cropland area, and urban area from the total area of each county.

Land-use maps for each year from 1850 to 1973 were reconstructed based on the recorded area fractions of different land-use types in each grid pixel of the 1-km resolution 1974 land-use map. We assigned one of the three land-use types (forest, urban, and cropland/pasture) to each nonwater grid pixel based on the area fraction of different land-use types in that grid pixel, so that the higher the area fraction of a land-use type in the grid pixel the higher the probability that kind of land-use type will be assigned to the grid pixel. For each land-use map, we controlled the total number of pixels that were assigned a certain land-use type, so that the total area of each land-use type would match the historical census data for that year.

Other Data Sets

In this study, climate, soil, elevation, and cloudiness are assumed to be stable annually. The elevation data represent a 1-km aggregation of the 7.5-min USGS National Elevation Dataset (<http://www.edcnts12.cr.usgs.gov/ned/ned.html>). We also used a 1-km resolution digital general soil association map (STATSGO map) developed by the US Department of Agriculture (USDA) Natural Resources Conservation Service to create a soil texture map of the study area. The texture information of each map unit was estimated using the USDA

soil texture triangle (Miller and White 1998). We used a 2002 six-category (coniferous forest, deciduous forest, mixture forest, cultivated land, intensive urban and transportation, and water) land-cover map (substituting the potential vegetation type of this region for the cultivated land and urban types) as the natural vegetation map of this region to generate baseline conditions in the equilibrium portion of the simulations. For the simulation, we used five major land-cover classes—urban impervious surfaces, coniferous forest, deciduous forest, mixture forest, and cropland/pasture. Lakes, streams, and other aquatic ecosystems were excluded from the simulation. We put the grassland category into the cropland/pasture category for this simulation, because natural grassland areas were small in this region.

Climate data were obtained from the National Climatic Data Center (NCDC). Monthly precipitation and average air temperature records (1970–2000) maintained by the 25 cooperative network stations in the three counties and the counties adjacent to the research region were used. For each month of the year, a temperature and precipitation raster layer at 1-km resolution was constructed using a Trend Surface Interpolation (TSI). Mean monthly cloudiness data for this study were derived from the Climatic Research Unit (CRU) TS 2.0 climate data set (Mitchell and others forthcoming). We then constructed the cloudiness map at 1-km resolution in the research area by linear interpolation.

RESULTS AND DISCUSSION

Land-cover Changes during 1974–2002

According to our results, from 1974 to 2002, abandoned cropland area in the three counties comprised 407 km², most of which was converted to forests (Table 2). At the same time, conversion to cropland and urban use had eliminated over 95 km² and 116 km² of forestland, respectively. The net effect was a slight increase in forest area from 2,414 km² in 1974 to 2,610 km² in 2002. Cropland area was reduced by more than 59%, whereas urban land use more than doubled. Forestland (including urban forest and woodlands) covered 78%, 81%, and 82% of the West GA counties in 1989, 1997, and 2000, respectively. These estimates were higher than the FIA figures of 74%, 74%, and 75% for these 3 years (<http://www.fia.fs.fed.us/rpa.htm>). This difference may be due to the fact that the areas identified as forest by remote sensing

Table 2. Land-cover Change between 1974 and 2002 (km²)

		Land Cover in 2002			Total in 1974
		Forest	Cropland/Pasture	Urban/Transportation	
Land Cover 1974	Forest	2,203	95	116	2,414
	Cropland/Pasture	407	169	68	644
	Urban/Transportation	0	0	48	48
Total in 2002		2,610	264	232	3,106

in our investigation included all the forest and woodland (including urban forests) that can be identified on the 30-m resolution images (80-m for Landsat MSS), whereas the FIA project estimates were based on a more restricted definition that requires a plot to have at least 10% tree stocks and to be at least 1 acre in size before it can be classified as forestland (USDA Forest Service 2005).

Total forest area changed little because land-use change due to cropland abandonment and urbanization tended to balance each other out. This finding is in agreement with the reports of other studies in the southeastern United States (Wear 2002; Milesi and others 2003). Most of the new development (63%) was due to the conversion of forests; a similar figure has been reported by the USDA Natural Resources Conservation Service (www.nrcs.usda.gov).

Rapid land-use change in this region has an impact on the age of forest stands (Harts 1980). Our data indicate that in 1989 about 63% of the forests were less than 40 years old. This value is a little lower than that given in FIA reports, which indicate that 75% of forests were in the 1–40 age class (<http://www.ncrs2.fs.fed.us/4801/fiadbf/>). Our older forest age structure may reflect the inability of remote-sensing sources to capture information on yearly forest management (such as rotation).

The major land-use change in Muscogee County was urbanization (Figure 2b) whereas the major land-use change in Meriwether County was cropland abandonment accompanied by considerable agricultural conversion (that is, cropland/pasture converted from forest). In Harris County, a large area of cropland abandonment was evident in the northwest part of the county, while urbanization characterized the southeast. Because of the military installation to the south, Columbus expanded in a northerly direction. In addition, the urbanization rate changed along the gradient. The urban area of Muscogee County, which had the highest percentage of urban (or impervious surface) area (6.9% of total county area in the mid-1970s), increased 3.31 km² annually, whereas the urban area of Meriwether County (which had the lowest

urban percentage, at 0.06% of total county area in the mid 1970s) increased only 0.93 km² each year. Harris County, located in the middle, gained 1.22 km² of urban land each year. Our analysis indicates that since 1990, urbanization has become more evident in all three west Georgia counties (Figure 3), such that impervious surface coverage (ISC) in West GA increased from 1.5% in 1974 to 7.5% in 2002. Our land-cover data set, however, may overestimate ISC in urban regions and slightly underestimate ISC in rural regions.

We selected eight West GA watersheds and compared their ISCs in our 2002 land-use map to the results of another study based on high-resolution color infrared (CIR) aerial photos (Lockaby and others 2005) taken in 2003. Our estimate of 0.4% ISC for the most undeveloped watershed is lower than the estimate of 0.7% ISC by Lockaby and others 2005; whereas our estimate of 49% ISC for the most developed watershed is higher than their estimate of 42% ISC. For the six watersheds whose ISC falls between 1.5% and 7.5%, our estimates of the ISC are 1.2%, 1.5%, 1.6%, 1.8%, 2.3%, and 3.5%, respectively; these figures generally agree with the estimates (1.5%, 1.6%, 1.8%, 1.9%, 2.5%, and 2.6%, respectively) given by Lockaby and others (2005).

According to Wear (2002), the two major changes in land use that occurred in the southern United States during the latter half of the 20th century were urbanization (most areas being converted from forest land) and reforestation on abandoned croplands. As a result of a balance between urbanization and reforestation, forest area in the region has remained roughly constant. Our results indicate that the pattern of land-use change at the West GA rural–urban interface matches well with the general pattern of land-use change observed in the southeastern United States as a whole.

In general, land cover in this area has changed significantly since the mid-1970s. Although total forest cover did not change much, disturbed forestlands over the 29-year period represent more than 27% of total forestland. Therefore, in some areas of the region, vigorous young forest growth

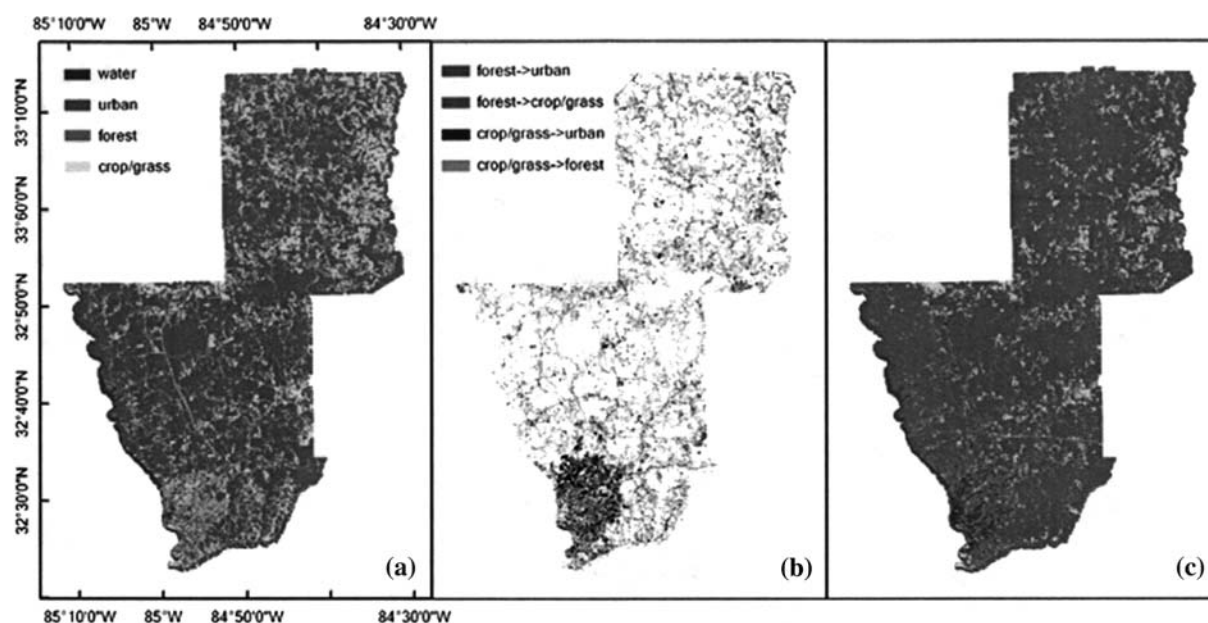


Figure 2. Land-cover change in three counties of west Georgia from the mid-1970s to the early 2000s. **a** Land-cover map in 1974. **b** Land-cover change (LCC) from 1974 to 2002. **c** Land-cover map in 2002.

on abandoned croplands sequestered carbon dioxide (CO_2), whereas, in other parts of the region, deforestation and urbanization released CO_2 . Clearly, the fact that the land-use change processes counteracted each other was important for the balance of C in the region.

Modeled Estimates of Carbon Storage and Net Primary Productivity

According to our simulation results, in 2002, total C storage in the West GA counties was 42 million ton (M t), including 27 M t stored in vegetation and 18 M t stored in soil. Vegetation C in the three counties of Meriwether, Harris, and Muscogee was 12 M t, 13 M t, and 2 M t, respectively. Soil organic C in the three counties was 8 M t, 8 M t, and 2 M t, respectively. The total ecosystem C density of the West GA region increased slightly from 13,409 g C m^{-2} in 1974 to 13,539 g C m^{-2} in 2002 (Table 3). Ecosystem storage of C was nearly equal in 1980 (13,543 g C m^{-2}) and 2002 (13,539 g C m^{-2}). In other words, total C storage did not change much during the study period. Changes in ecosystem C storage induced by land-use change varied spatially (Figure 4). Two C-sink regions located in the northeast and southeast (Fort Benning) corners reflected areas that were less influenced by urbanization. The most intensive C loss occurred at the periphery of urban areas at the urban-rural interface. Both Imhoff and others (2000) and Milesi and others (2003) observed that most of the newly

developed land is located at the periphery of large urban areas. Our results also indicate that large C sources emerge at rural-urban interfaces.

Model simulations show that during the 1990s the coniferous forest C stocks averaged 6,755 g C m^{-2} . This estimate was higher than the eastern softwood average biomass of 5,500 g C m^{-2} derived from FIA inventory data (Brown and Schroeder 1999; Brown and others 1999). One possible explanation for our higher estimate is that we included all vegetative biomass in our simulation, whereas the estimate derived by Brown and others (1999) included only trees with a diameter of more than 2.54 cm at breast height. Our estimate of vegetation C in deciduous forests is 14,352 g C m^{-2} , well within the range of eastern hardwood C density (11,800–17,200 g C m^{-2}) documented in the FIA data (Brown and others 1999). The average upper 1 m of soil C density was about 6,500 g C m^{-2} in the 1990s. That figure is comparable to Birdsey and Lewis's (2002) estimate of 7,140 g C m^{-2} for Georgia forest soils in 1997. The soil C density of deciduous forests in the area is 9,347 g C m^{-2} , which is close to Turner and others (1995) estimate of 9,500 g C m^{-2} in the eastern United States.

Our simulated average 1990s NPP for coniferous forests was 494 $\text{g C m}^{-2} \text{ y}^{-1}$. The predominant coniferous forest type in the research region was loblolly pine (*Pinus taeda* L.) (Thompson and Thompson 2002). Teskey and others (1987) reported that aboveground NPP for loblolly pine forests ranged between 100 and 500 $\text{g C m}^{-2} \text{ y}^{-1}$ and

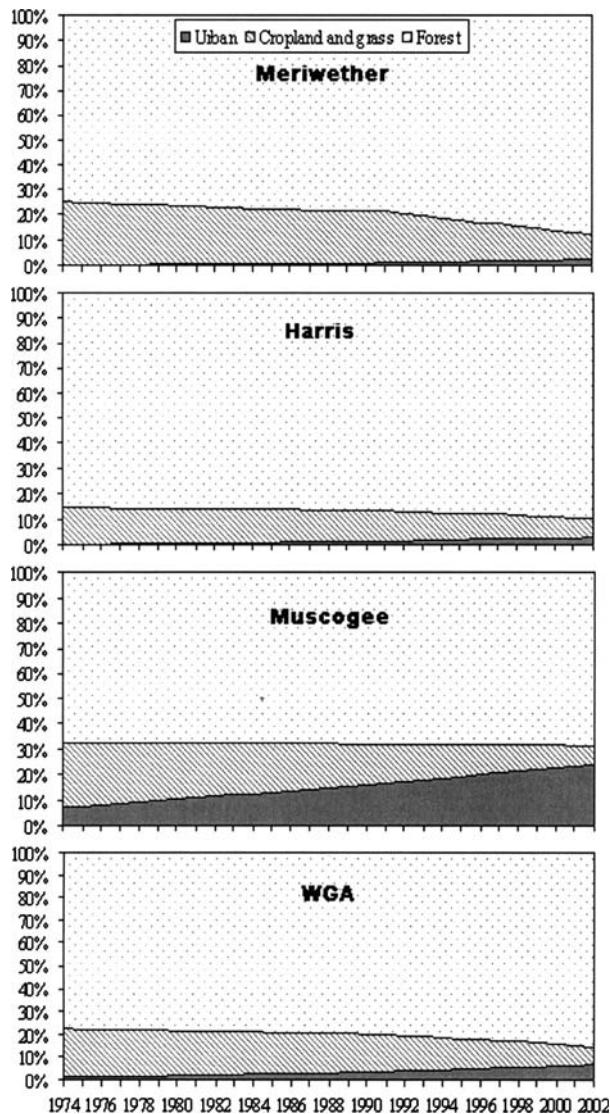


Figure 3. Land-cover change in three counties of west Georgia during the 29-year period from 1974 to 2002.

averaged $300 \text{ g C m}^{-2} \text{ y}^{-1}$. If belowground production equals 40% of aboveground NPP (Nadelhoffer and others 1985; McNulty and others 1994), the total NPP will be $420 \text{ g C m}^{-2} \text{ y}^{-1}$. This value is very close to our estimate for total NPP ($494 \text{ g C m}^{-2} \text{ y}^{-1}$). Our simulated average 1990s NPP for deciduous forest was $1,020 \text{ g C m}^{-2} \text{ y}^{-1}$. This figure is similar to the one reported by Melisi and others (2003) ($1,081 \text{ g C m}^{-2}$ in Georgia) and also falls well within the range of $805\text{--}1,715 \text{ g C m}^{-2} \text{ y}^{-1}$ derived by Brown and Schroeder (1999).

Effects of Land-use Change on Ecosystem Carbon Storage

Our simulation indicates that from 1974 to 2002 reforestation in West GA sequestered about

$23.0 \text{ g C m}^{-2} \text{ y}^{-1}$ which was offset by the $18.4 \text{ g C m}^{-2} \text{ y}^{-1}$ released by deforestation. The net C exchange was an uptake of about $4.6 \text{ g C m}^{-2} \text{ y}^{-1}$ by terrestrial ecosystems. Although the magnitude was similar, the spatial patterns of C fluxes induced by deforestation and reforestation were quite different. As shown in Figure 4, the C losses occurred in only a few locations in the study area. Although, concentrated areas resulted in large C losses, the spatial pattern of C gains was diffuse and its magnitude in each grid was relatively small. The negative NCE induced by deforestation was much more variable than the positive NCE due to forest regrowth (Figure 5). Our analysis showed that the release of C due to deforestation generally followed the patterns of annual deforestation, whereas C sequestration due to reforestation did not match the patterns of annual reforestation (that is abandoned croplands). The differing patterns between deforestation NCE and reforestation NCE may be a reflection of the differences of scale for the two processes. Although, ecosystem C loss responds instantly to deforestation events, C sequestration due to forest regrowth is a relatively slow and stable process. Furthermore, newly reforested areas comprise only a small part of the growing forests that are sequestering C. Therefore, reforestation, unlike deforestation, has a long lag effect. As shown in Figure 5, the short-term period fluctuation in total NCE was dominated by deforestation, whereas the long-term trend showed that a net accumulation of C resulted from reforestation. The combined effects of these two kinds of processes at different scales generated a complex NCE pattern.

We further identified the effects of different deforestation processes on the regional C balance. During the study period, agricultural deforestation was the major C-release process. Agricultural deforestation released -15.6 g C m^{-2} per year—4.6 times more than the C released by urbanization. However, the relative importance of these two processes changed both along the rural-urban gradient and through time (Figure 5). In Meriwether County, which is furthest from Columbus, agricultural deforestation dominated the negative NCE fluxes, and the effect of urbanization was negligible. Conversely, in Muscogee County, which is within the Columbus city limits, the negative NCE generated by urbanization is 2% larger than that generated by agricultural deforestation. In Harris County, agricultural deforestation released 25 times more C each year than urbanization before 1998. In the last 2 years of the 1990s, C released from agricultural deforestation declined to about $-2.7 \text{ g C m}^{-2} \text{ y}^{-1}$, but this figure was still higher

Table 3. Carbon (C) Density in Three West Georgia Counties (1974–2002) (g/m^2)

Year	1974	1980	1990	2000	2002
Vegetation C density	7,922	8,051	8,004	8,117	8,136
Soil C density	5,487	5,492	5,518	5,414	5,403
Total ecosystem C density	13,409	13,543	13,522	13,531	13,539

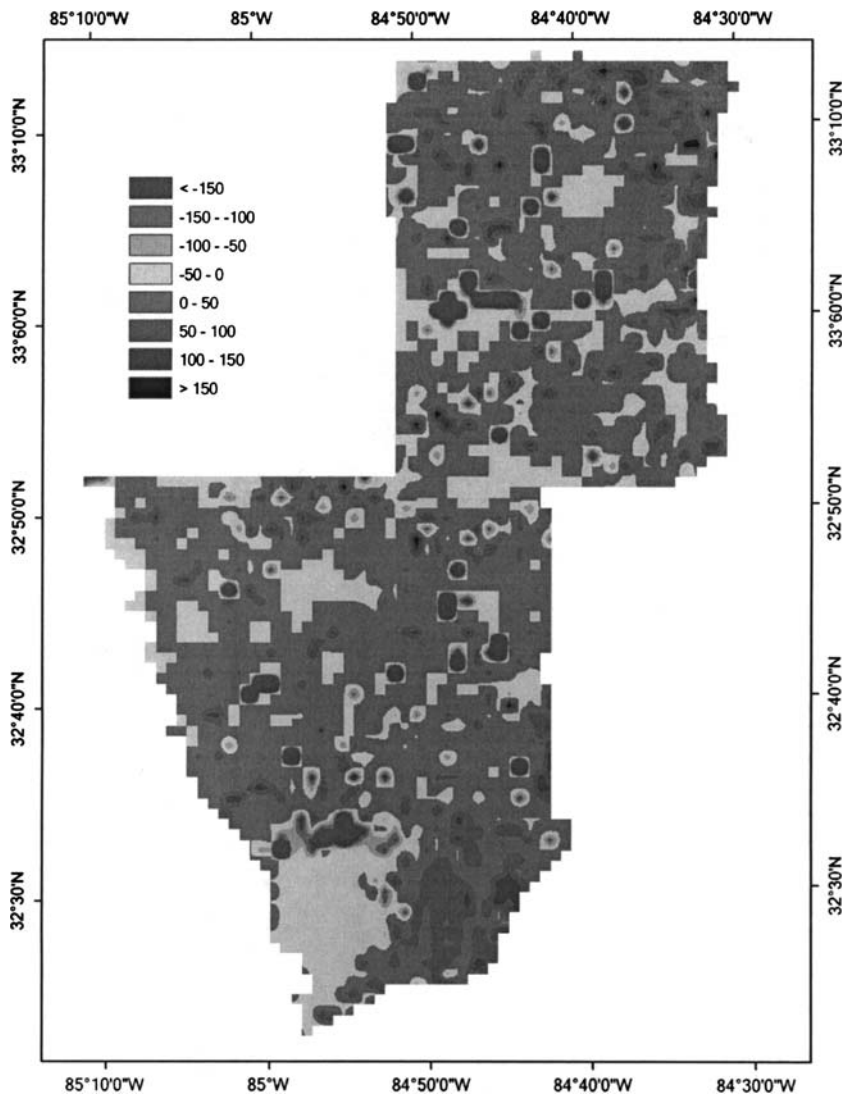


Figure 4. Net carbon (C) exchange between the atmosphere and terrestrial ecosystems in three west Georgia counties from 1974 to 2002 ($\text{g C m}^{-2} \text{y}^{-1}$). Positive value means C sink; negative value means C source.

than the $-1.4 \text{ g C m}^{-2} \text{y}^{-1}$ released by urbanization. After 2000, the effect of agricultural deforestation was only 63% of the effect of urbanization.

For the entire West GA research region, the importance of agricultural deforestation to the ecosystem C balance also changed through time. In the 1980s, agricultural deforestation generated considerable negative NCE fluxes (approximately $-24 \text{ g C m}^{-2} \text{y}^{-1}$)—about 14 times the amount released by urbanization ($-1.7 \text{ g C m}^{-2} \text{y}^{-1}$). This flux was greater than the $22 \text{ g C m}^{-2} \text{y}^{-1}$ NCE sequestered by forest regrowth, making the West GA area

a net C source. In the 1990s, the amount of C released by urbanization increased to $-3.4 \text{ g C m}^{-2} \text{y}^{-1}$ whereas the amount of negative NCE due to agricultural deforestation declined to about $-12 \text{ g C m}^{-2} \text{y}^{-1}$ —only 50% of the amount sequestered in the 1980s. During this decade, after the reduction in cropland area (Figure 2), the amount of C release due to agricultural deforestation decreased by 50% (from $16 \text{ g C m}^{-2} \text{y}^{-1}$ in the first half of the 1990s to $8 \text{ g C m}^{-2} \text{y}^{-1}$ in the late 1990s). However, agricultural deforestation still accounted for more than 70% of the total C

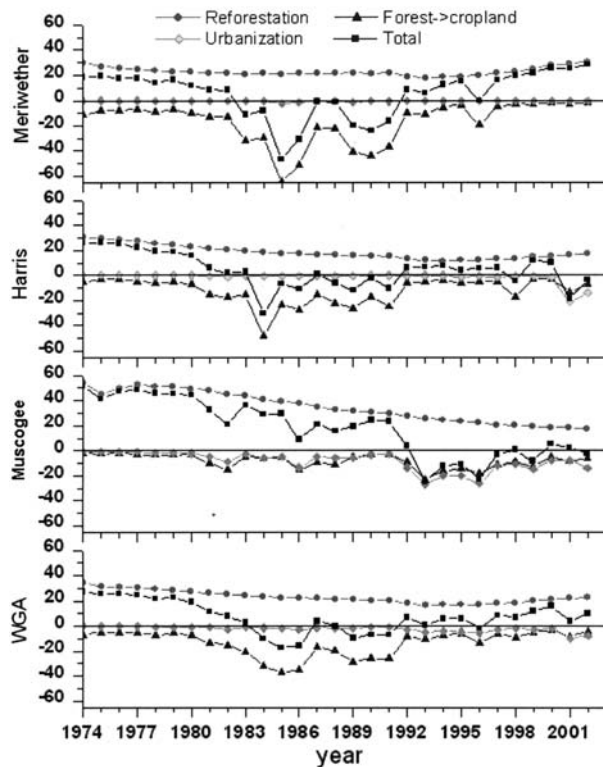


Figure 5. Relative contribution of different land conversions to net carbon exchange (NCE) (g/m^2) between terrestrial ecosystems and the atmosphere, as estimated by TEM. Positive NCE means C sink; negative NCE means C source.

released by deforestation in the 1990s. As a result of reduced deforestation and steady forest regrowth, West GA became a net C sink of $3 \text{ g C m}^{-2} \text{ y}^{-1}$ in the 1990s. In the early 2000s, the negative NCE due to agricultural deforestation decreased to $-5 \text{ g C m}^{-2} \text{ y}^{-1}$ about 40% of the average NCE in the 1990s—While the negative NCE due to urbanization in the early 2005 was nearly three times the urbanization NCE in the 1990s. As a result, urbanization released more C than agricultural deforestation. However, the total C released by deforestation was only 55% of the C sequestered by forest regrowth, and reforestation has dominated the West GA C balance since that time.

Due to the uncertainty associated with models and data, estimates of the terrestrial C balance remain problematical. This study provides several valuable lessons that may help to reduce that uncertainty by improving our ability to simulate the effects of land-use change on ecosystem C dynamics within the southeastern United States. First, our study in West GA shows that historical land-use change has strong legacy effects on

ecosystem C cycles. Recognition of this effect must be incorporated into any attempt to simulate long-term and large-scale processes such as regional forest regrowth. Therefore, accurate historical land-use maps need to be reconstructed before we can understand the response of southeastern US ecosystems to land-use change. Accurate assessment of regional C budgets in the Southeast must await spatially explicit reliable data sets that include more comprehensive land-use data for the entire region (McNulty and others 1994; Wofsy and Harris 2002). Second a fuller consideration of the large-scale land-use change that the Southeast has undergone the last several decades is imperative. Our results show that the relative roles of major land-use change processes on C storage have changed over time. By now urbanization has become a significant regional C-release process. A similar pattern has been predicted for the entire southeastern United States by other investigators (Wear 2002). Therefore, any successful simulation of land use as it relates to the C dynamics of this region will depend on the capacity to stimulate urban ecosystem processes. Finally, because forest NPP plays an important role in the C balance of West GA and forest plantations are expanding (Thompson and Thompson 2002; Wear 2002), current biogeochemical models need to be improved such that they better represent the structure and dynamics of managed immature forests (Song and Woodcock 2003) before we can successfully assess the C fluxes in the Southeast.

ACKNOWLEDGMENT

This study was supported by the Auburn University Peak of Excellence Program, the EPA STAR program, the McIntire-Stennis Program, and the USDA Forest Service. We thank David Kicklighter for helpful discussion in the early stages of this study. We are also grateful to Dr. Christine Goodale, Dr. Hua Chen, and two anonymous reviewers for their critical comments on the manuscript.

REFERENCES

- Arnold CL, Gibbons CJ. 1996. Impervious surface coverage: the emergence of a key environmental indicator. *J Am Plan Assoc* 62:243–58.
- Birdsey RA, Lewis BM. 2002. Carbon in U.S. Forests and Wood Products, 1987–1997: state-by-state estimates. General Technical Report NE-310. Washington (DC): US Department of Agriculture Forest Service. 47 p.
- Bousquet P, Peylin P, Ciais P, Le Quere C, Friedlingstein P, Tans PP. 2000. Regional changes in carbon dioxide fluxes of land and oceans since 1980. *Science* 290:1342–6.

- Brown SL, Schroeder PE. 1999. Spatial patterns of aboveground production and mortality of woody biomass for eastern US forests. *Ecol Appl* 9:968–80.
- Brown SL, Schroeder PE, Kern JS. 1999. Spatial distribution of biomass in forests of the eastern USA. *For Ecol Manage* 123:81–90.
- Fan S, Gloor M, Mahlman J, Pacala S, Sarmiento J, Takahashi T, Tans P. 1998. A large terrestrial carbon sink in North America implied by atmospheric and oceanic carbon dioxide data and models. *Science* 282:442–6.
- Felzer B, Kicklighter DW, Melillo JM, Wang C, Zhuang Q, Prinn R. 2004. Effects of ozone on net primary production and carbon sequestration in the conterminous United States using a biogeochemistry model. *Tellus* 56B:230–48.
- Goodale CL, Apps MJ, Birdsey RA, Field CB, Heath LS, Houghton RA, Jenkins JC, and others 2002. Forest carbon sinks in the Northern Hemisphere. *Ecol Appl* 12:891–9.
- Guo LB, Gifford RM. 2002. Soil carbon stocks and land use change: a meta analysis. *Global Change Biol* 8:345–60.
- Hart JF. 1980. Land use change in a piedmont county. *Ann Assoc Am Geogr* 70:492–525.
- Houghton RA. 1999. The annual net flux of carbon to the atmosphere from changes in land use 1850–1990. *Tellus* 51B:298–313.
- Houghton RA, Hackler JL, Lawrence KT. 1999. The US carbon budget: contributions from land-use change. *Science* 285:574–8.
- Imhoff ML, Tucker CJ, Lawrence WT, Stutzer DC. 2000. The use of multisource satellite and geospatial data to study the effect of urbanization on primary productivity in the United States. *IEEE Trans Geosci Remote Sens* 38:2549–56.
- Jensen JR, Ed. 1996. *Introductory digital image processing*. 2nd ed. Prentice-Hall Upper saddle River, New Jersey.
- Lockaby BG, Zhang D, McDaniel J, Tian H, Pan S. 2005. Interdisciplinary research at the urban–rural interface: the Westga Project. *Urban Ecosyst* 8:7–21.
- McGuire AD, Sitch S, Clein JS, Dargaville R, Esser G, Foley J, Heimann M, and others 2001. Carbon balance of the terrestrial biosphere in the twentieth century: analyses of CO₂, climate and land use effects with four process-based ecosystem models. *Global Biogeochem Cycles* 15:183–206.
- McNulty SG, Vose JM, Swank WT, Aber JD, Federer CA. 1994. Regional-scale forest ecosystem modeling: database development, model predictions and validation using a Geographic Information System. *Clim Res* 4:223–31.
- Milesi C, Elvidge CD, Nemani RR, Runing SW. 2003. Assessing the impact of urban land development on net primary productivity in the southeastern United States. *Remote Sens Environ* 86:401–10.
- Miller DA, White RA. 1998. A conterminous United States multi-layer soil characteristics data set for regional climate and hydrology modeling. *Earth interactions* 2: Available Online at: <http://www.EarthInteractions.org>.
- Mitchell TD, Jones PD (2005) An improved method of constructing a database of monthly climate observations and associated high-resolution grids. *Int. J. Climatol.* 25: 693–712.
- Nadelhoffer K, Aber JD, Melillo JM. 1985. Fine roots, net primary production, and soil nitrogen availability: a new hypothesis. *Ecology* 66:1377–90.
- Pacala SW, Hurtt GC, Baker D, Peylin P, Houghton RA, Birdsey RA, Heath L, and others 2001. Consistent land- and atmosphere-based U.S. carbon sink estimates. *Science* 292:2316–1320.
- Schimel D, Melillo JM, Tian H, McGuire AD, Kicklighter DW, Kittel T, Rosenbloom N, and others 2000. Contribution of increasing CO₂ and climate to carbon storage by ecosystems in the United States. *Science* 287:2004–6.
- Song C, Woodcock CE. 2003. A regional forest ecosystem carbon budget model: impacts of forest age structure and land-use history. *Ecol Model* 164:33–47.
- Teskey RO, Bongarten BC, Clegg BM, Dougherty PM, Hennessey TC. 1987. Physiology and genetics of tree growth response to moisture and temperature stress: an examination of the characteristics of loblolly pine (*Pinus taeda* L.). *Tree Physiol* 3:41–61.
- Thompson MT, Thompson LW. 2002. Georgia's forests, 1997. Resource Bulletin SRS-72. Southern Research Station, US Department of Agriculture, Forest Service. Available online at: <http://www.treesearch.fs.fed.us/pubs/srs/>.
- Tian H, Melillo JM, Kicklighter DW, McGuire AD, Helfrich JVK III, Moore B III, Vörösmarty CJ. 1998. Effect of interannual climate variability on carbon storage in Amazonian ecosystems. *Nature* 396:664–7.
- Tian H, Melillo JM, Kicklighter DW, McGuire AD, Helfrich J. 1999. The sensitivity of terrestrial carbon storage to historical atmospheric CO₂ and climate variability in the United States. *Tellus* 51B:414–52.
- Tian H, Melillo JM, Kicklighter DW, Pan S, Liu J, McGuire AD, Moore B III. 2003. Regional carbon dynamics in monsoon Asia and its implications for the global carbon cycle. *Global Planet Change* 37:201–17.
- Turner DP, Koerper GJ, Harmon ME, Lee JJ. 1995. A carbon budget for forests of the conterminous United States. *Ecol Appl* 5:421–36.
- USDA Forest Service. 2005. Forest inventory and analysis national core field guide: ver 3: Vol I. Available online at: <http://www.fia.fs.fed.us/library/field-guides-methods-proc/>.
- Waisanen PJ, Bliss N 2002. Changes in population and agricultural land in conterminous United States counties, 1790 to 1997. *Global Biogeochem Cycles* 16:1137–1156.
- Wear DN. 2002. Land use. In: Wear DN, Greis JG, Eds. *Southern forest resource assessment final report*. Available online at: <http://www.srs.fs.usda.gov/sustain/report/>.
- Wofsy SC, Harriss RC. 2002. The North American Carbon Program (NACP). Report of the NACP Committee of the US. Interagency Carbon Cycle Science Program. Washington (DC): US Global Change Research Program.