Forest Carbon Dynamics Associated with Growth and Disturbances in Oklahoma and Texas, 1992–2006

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ABSTRACT

Quantifying forest carbon changes associated with growth and major disturbances is important for management of greenhouse gas emissions related to forests. Regional-level approaches with improved local growth data may refine estimates obtained using coarser resolution information. This study integrates remote-sensing-derived land cover change products, harvest data, forest fire data, and local forest growth estimates at the county level to identify forest ecosystem carbon change for the states of Oklahoma and Texas (1992–2006). Whereas Oklahoma was a carbon sink of 0.5 Tg C yr^{-1}, Texas was estimated to be a carbon source of -1.8 Tg C yr^{-1} for the period. The two states together functioned as a carbon source of -1.3 Tg C yr^{-1} for the entire period, although it was a small sink of 0.1 Tg C yr^{-1} in the recent period of 2001–2006 due to reduced annual rates of net forest-to-nonforest conversion and harvesting, compared to those in the early period of 1992–2001. Most counties located in the western portions of both states were small sinks of carbon during the period. Even though their growth rates are greater, many counties in the eastern portions of both states were carbon sources due to a higher intensity of forest-related disturbances. A sensitivity analysis was conducted to investigate possible double-counting of harvest and cover change by assuming half of the sequestration and emissions from land cover changes were already counted as harvest. Results indicated Oklahoma would be a sink of 1.0 Tg C yr^{-1}, and Texas would be a small carbon source of -0.1 Tg C yr^{-1}. Uncertainty in forest area for the western portions of these states remains an important source of potential error.

Keywords: carbon sequestration, carbon emission, net carbon exchange, ecological region, major disturbances

Forest ecosystems play an important role in the global carbon cycle. Spatially explicit estimation of carbon sinks and sources can enhance our understanding of the balance between forest growth and disturbances and can improve our ability to better manage forest ecosystems for reducing greenhouse gas emissions and mitigating climate change. Several types of methodologies (Zheng et al. 2003), including resource inventory data, models, and remote sensing observations, have been used to quantify forest carbon dynamics in the United States. Increasing availability of remote sensing (RS) data and the rapid development of RS techniques in recent decades have provided a relatively consistent, reliable, efficient, and practical means for studying terrestrial ecosystems at multiple scales and at periodic intervals (Hansen et al. 2000, Zheng et al. 2004, Xian et al. 2009).

Previous assessments of methodologies have indicated that different approaches using RS-based gross land cover change and inventory-based net change relating to nonforest becoming forest, forest becoming nonforest, and forest management have different limitations and advantages (Drummond and Loveland 2010, Hansen et al. 2010, Reams et al. 2010). Land cover data sets provide limited information about carbon stocks but can identify areas of disturbance, whereas inventory data provide useful information for carbon stock or stock-change calculations but have limited information for quantifying disturbance effects, especially when inferences are of interest over relatively small areas such as counties (Green et al. 1987). Zheng et al. (2011) demonstrated a new methodology to estimate forest ecosystem carbon change in the conterminous United States between 1992 and 2001 based on a land cover change map and carbon density changes based on USDA Forest Inventory and Analysis (FIA) field plot data, coupled with data sets of differing spatial resolution on disturbances such as harvest and forest fires. Similarly, Zhang et al. (2012) developed an approach that integrated remote sensing, inventory data, and a process model to assess the effects of disturbance and nondisturbance factors on annual changes in carbon stocks at the 1-km^2 spatial resolution for the conterminous United States.

It has been commonly recognized that US forests have functioned as an overall carbon sink in recent decades (Pacala et al. 2001, Woodbury et al. 2007, Heath et al. 2011). But reported regional patterns in terms of carbon sinks or sources across the country differ due to variations in methods, scales of observation, and data limitations (Turner et al. 1995, Potter et al. 2007, Zheng et al. 2011). A

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This article uses metric units; the applicable conversion factors are: meters (m): 1 m = 3.3 ft; kilograms (kg): 1 kg = 2.2 lb; megagrams (Mg): 1 Mg = 2,204.6 lb; hectares (ha): 1 ha = 2.47 ac.

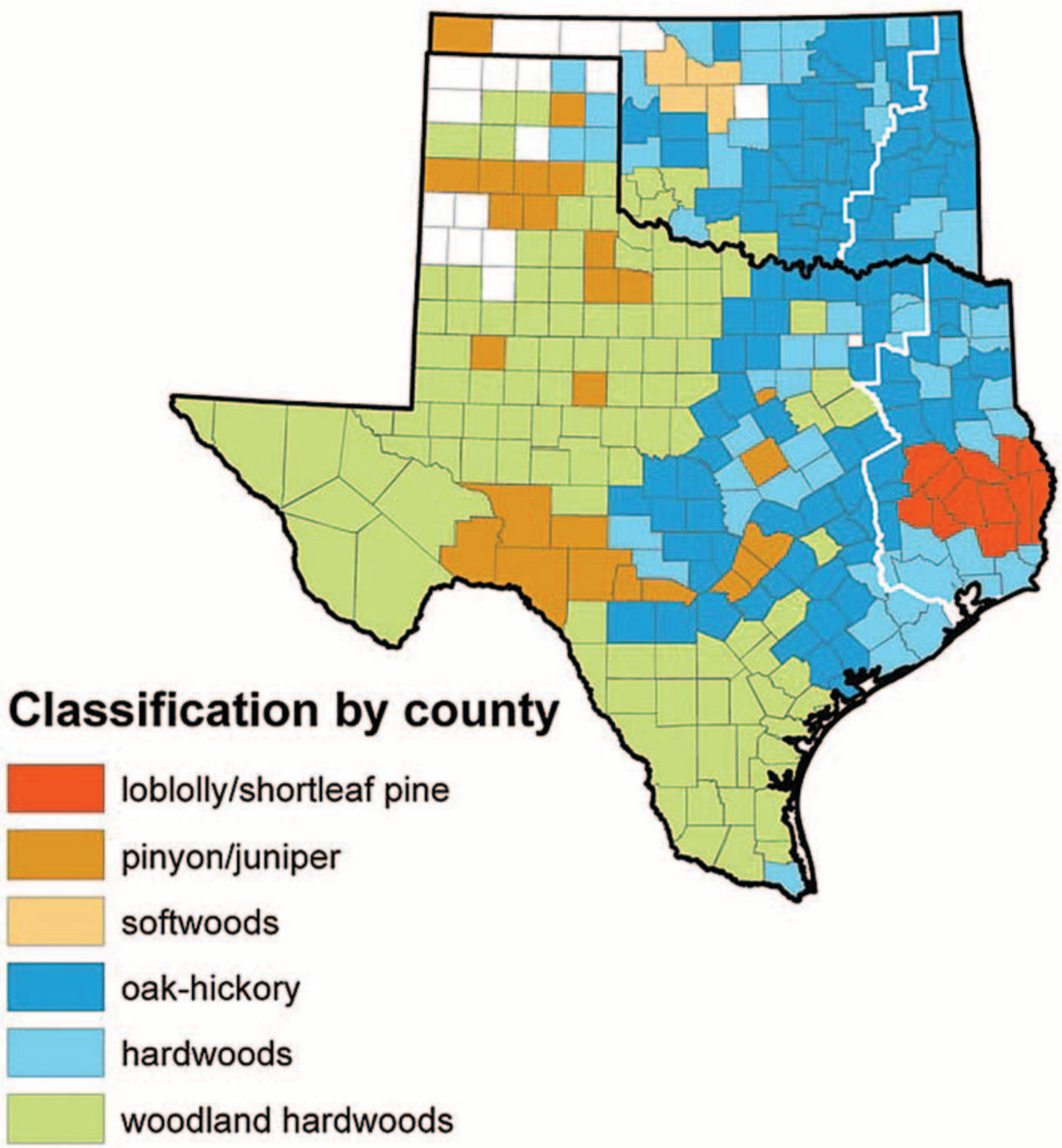

Figure 1. Spatial distribution of the most-common forest type group at the county level in the states of Oklahoma (upper state delineated by dark lines) and Texas. Counties in white within the state boundaries indicate no forestland. White lines are divisions between East (characterized by high forest productivity and greater disturbances) and C&W inventory surveys (USDA Forest Service 2011).

previous national-level study was limited in its ability to provide estimates for the states of Oklahoma and Texas in particular because the methodology used did not take into account the lack of forest inventory data in the states' central and western portions (Zheng et al. 2011). It has been shown that the forest area estimates in Texas are sensitive to forestland definitional changes (Coulston et al. 2010). However, the data sources in our approach are consistent over the period and also may be used to estimate changes that can be attributed directly to types of disturbance. The specific objectives of this study are to: (1) provide a complete current estimate of forest ecosystem carbon change in Oklahoma and Texas and compare the results between this study and a previous study; (2) compare the carbon dynamics associated with disturbances between the 9-year period of 1992–2001 and the 5-year period of 2001–2006; and (3) explore spatial patterns and differences in terms of net forest carbon change at the county scale and between the central and western and eastern portions of the states.

Materials and Methods

Study Area

Our study area includes the entire states of Oklahoma and Texas in the south central United States totaling 854,800 km^2 (excluding inland water), with Oklahoma comprising 21% of the total area. Thirteen percent of the total two-state area was defined as forested in 2006 based on RS-derived observations (Xian et al. 2009). These two states are unique because their territories straddle traditional division lines between the eastern and western United States in terms of ecology and timber productivity (Woudenberg et al. 2010) (Figure 1). Inventory-based characterizations of forests in the western portion of the two states suggest both lower productivity and fewer disturbances in comparison with eastern forests. Historically, FIA inventories have focused on the more productive eastern forests (USDA Forest Service 2011), but survey data are now available for the entire portion of both states (hereafter, the eastern and central-and-western survey areas are referred to as East and C&W, respectively). Dominant forest types in the central portions of the states are oak-hickory types, while types further west are predominantly mesquite woodlands and juniper woodlands of generally low productivity (Figure 1). The eastern portions of the states are predominantly oak-dominated hardwood forest or loblolly-pine-dominated softwood forest. Collectively, these forest types account for over 90% of forestland in each of the East and C&W surveys.

Data Sets

Our general approach follows that of Zheng et al. (2011), in which forest carbon dynamics for a given period were calculated based on the areas for different forest cover changes (see definitions below) identified from RS observations, forest growth rates (carbon per ha per year) obtained from field inventory data, and emissions from harvesting data and wildfires. We describe these four data sets, emphasizing the updates and differences from the previous national-level study.

Land Cover Change Assessments

Two land cover change maps from the US Geological Survey (USGS) National Land Cover Data sets (NLCD) derived from 30-m Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) satellite data were used: (1) the NLCD change map of 1992–2001 (Fry et al. 2009) and (2) the NLCD change map of 2001–2006 (Xian et al. 2009). Both maps provided spatial land cover change information about conversion from one land cover to another, or no change in land cover, for the given periods. The NLCD 2001–2006 change map was designed in such a way that continual monitoring of the nation's land cover change can be achieved.

We performed a consistent analysis at Anderson Level I (Anderson et al. 1976) for the entire period of 1992–2006 and for the two time periods separately after, aggregating the change detections for 2001 to 2006 from Level II to Level I. As a result, land cover types with a single-digit coding indicated there was no change in cover type for a given land during the study period. The classes are: (1) open water, (2) urban, (3) barren, (4) forest (including deciduous, evergreen, and mixed), (5) grass/shrub (G/S), (6) agriculture, and (7) wetland. Any classes of two-digit coding represented land cover change from one type to another during the period. For example, class 42 meant that land was converted from forest in the beginning year to urban in the ending year. We aggregated forest-related land cover changes during the study period into three categories: (1) nonforest to forest cover change (NFCC) (including change classes of 14, 24, 34, 54, 64, and 74), (2) forest to nonforest cover change (FNCC) (including change classes of 41, 42, 43, 45, 46, and 47), and (3) forest remaining forest (class of 4).

We note that "FNCC" as used here is a land cover designation, not a land use designation. Thus, some cover changes are identified in this component that do not constitute land use change, such as some harvesting of older forests resulting in young forests. Furthermore, the definition of forestland is crucial for forest area estimation because different data sets are often based on different definitions. The RS-based area estimation used here differs from the typical FIA-based area estimation, which has been a land use definition and includes a minimum productivity threshold as well as explicitly identifies plant species as trees or shrubs regardless of height. These differences in definitions can complicate the carbon estimates and comparisons of approaches. However, as noted in Smith et al. (2009), FIA used RS-based area estimates in west Texas and western Oklahoma for its forest statistics at that time and in all reports previous to that time, so the 2006 forest area for the states are similar to ours.

Throughout this analysis, we use the terminology of FNCC instead of "deforestation" because the change may not be deforestation, which is a land use conversion from forest to nonforest. Similarly, we used "NFCC" for all cover changes from nonforest to forest in this study. However, the results from this study for the eastern portion of the two states during the period of 1992–2001 should be comparable with the results for the corresponding area from a previous study (e.g., Zheng et al. 2011). At that time of that study, forest growth data were unavailable in the C&W counties of the two states.

Harvest Data and Carbon Associated with Harvest

Carbon change associated with forest harvest was based on FIA inventory and timber product output (TPO) data. We estimated carbon stored in use and landfill at 100 years following harvest by using information for individual years during the period of 1992–2006 when data were available. Estimates of carbon in harvested wood are based on roundwood volumes in Table 5 of USDA Forest Service (2012) using the 100-year method (Miner 2006) and carbon allocation factors for harvested wood (Smith et al. 2006). For comparison purposes, carbon emissions from harvest used in this study were adjusted to include logging residue left in the forest at the county level by using an empirical regression model ($r^2 = 0.996$, data not shown) based on information from a previous study (Zheng et al. 2011). This residue amount was assumed to be emitted immediately, although in reality it will decay and be emitted over time so we are overestimating emissions during the period slightly. Once the amount of average annual carbon emission from harvest during the study period was calculated (carbon removal in wood, branches, and leaves through harvests minus harvested wood but stored in use and landfill), it was multiplied by the numbers of years for the corresponding study periods to estimate the total emissions associated with harvest over the NLCD change intervals mentioned above. Harvest data are only currently collected by FIA for the 61 counties in East Oklahoma (18) and Texas (43), but commercial harvest in the C&W portion of the region is estimated to be minor in comparison. In Texas, less than 2% of the total industrial harvest in the state in 2009 is outside these counties (Li et al. 2010), and Oklahoma likely has similar statistics (Johnson et al. 2010).

Because there were no specific county-level harvest statistics on wood removed in C&W regions and because less than 2% was estimated to be outside the East, we set the carbon removals from counties located in C&W Oklahoma and Texas to zero for this analysis. Average numbers of years available for harvest data for the counties during the entire study period were 3.4 (varying from 1 to 4 years) in Oklahoma and close to seven (varying from five to seven years) in Texas, respectively. Because of the possible double-counting of areas of harvest and FNCC, we further conducted a sensitivity analysis using the results from the above data analysis but modified the results for the East regions assuming that half the emissions from harvesting were already accounted for as FNCC.

Localized Net Forest Growth Rates for the Study Area

Net forest growth was calculated according to methods of Zheng et al. (2011), with the underlying forest growth estimates based on net carbon accumulation curves from Smith et al. (2006). Briefly, Smith et al. (2006) employ the traditional approach (e.g., see Mills and Kincaid 1992) of using data measured at one time arranged by volume or age, constructing a yield curve, and then taking the difference for growth estimates. However, forest types covering a significant portion of the C&W survey area (Figure 1) are not included in Smith et al. (2006) because no data were available. The recent establishment of the initial FIA forest inventory survey in C&W Oklahoma and Texas made it possible to develop net growth tables for the woodland forest types. Because total forestland was well represented by only four forest type groups, we developed a full set of new tables to maintain consistency among forestlands within this study. The forest type groups were loblolly/shortleaf pine, pinyon/juniper, oak/hickory, and woodland hardwoods (Woudenberg et al. 2010). We developed eight forest net growth tables in total for these four types, one set for NFCC and the other one for forest remaining forest. The new set of forest carbon tables provided plot-level carbon stock estimates by carbon pool according to stand age following Smith et al. (2006), but the estimates were based on more recent inventory data (USDA Forest Service 2011) and tree growth and yield were based on plot data fit to the von Bertalanffy growth model (Zeide 1993, Lei and Zhang 2004).

Application of these tables follows Zheng et al. (2011), where the first step was to characterize a representative forest stand for each county according to forest type and age. This determined the appropriate forest type and record (stand age) within the table to assign current carbon stock. Counties that were majority hardwood but not woodland hardwoods used the oak/hickory table; these were most commonly elm/ash/cottonwood. Counties that were majority softwood but not majority loblolly/shortleaf pine or pinyon/juniper only occurred in four C&W Oklahoma counties (as Eastern redcedar) so we used the pinyon/juniper table because it best matched the forest composition. Net annual carbon growth was based on the difference between carbon stocks at the age initially assigned for a county and the age at the end of the interval as well as the area expansion appropriate for the forest cover class.

Forest Fire Data and Emissions Calculations

Spatial fire maps from 1992 to 2006 for Oklahoma and Texas, compiled by the Monitoring Trends in Burn Severity (MTBS) project, were used to identify the location, extent, and burn severity of forest fires occurring during the study period (Brad Quayle, USDA Forest Service, pers. comm., May 10, 2012). These maps documented large fire events (greater than 202 ha and 404 ha in the eastern and western United States, respectively) from 1984 to present occurring on federal, state, and private lands using Landsat TM and ETM+ imagery (Eidenshink et al. 2007).

We compiled maps to generate two data sets for calculating annual mean carbon emissions from fires: (1) burn/reburn frequency (*BurnFreq*)—the number of times a given forested pixel burned during the study period; and (2) maximum burn severity (*MaxSev*)—the maximum recorded burn severity for a given forested pixel during the study period.

Carbon emissions from forest fires were estimated based on burned areas (including areas within fire perimeters that did not burn, which were classified as very low severity category in the fire data), carbon density, and carbon consumption rates following the method of Chen et al. (2011) given in Equation 1, varying substantially with burn severity (i)

$$\textit{Fire emission}_i = \text{Area burned}_i * \text{Carbon density} * \text{Proportion emitted}_i \quad (1)$$

Mean nonsoil forest C density data at the county level calculated according to methods of Smith et al. (2010) were obtained from FIA inventory data (USDA Forest Service 2011). Burn severity in the MTBS was categorized as one of four categories: (1) very low, (2) low, (3) moderate, and (4) high. Emission proportions for estimating carbon emissions from fires were set at 0.07, 0.20, 0.40, and 0.60 of the mean nonsoil C density, respectively, for the corresponding severity classes (Chen et al. 2011) of burned areas. In each pixel, the maximum burn severity was recorded and the corresponding C emission rate was used in the first-round calculation using Equation 1. For those areas that were burned more than once, we repeated the calculation using Equation 1, but we used a lower C density due to the expected reduction in C density caused by the previous fire, along with a constant C emission rate of 0.2 (Zheng et al. 2013). For example, if an area of 100 ha in size having C density of 50 Mg per ha, was burned twice during the study period with the maximum burn severity of moderate (C emission rate of 0.4), then the amount of C emitted from the first fire was $100 \times 50 \times 0.4 = 2{,}000$ Mg. The emitted C from the second fire was $100 \times 30 \times 0.2 = 600$ Mg. Thus, the total C emission for the area was 2,600 Mg (Zheng et al. 2013). The maximum burn frequencies included in carbon calculations were three for Oklahoma and two for Texas, which covered almost 100% of burned areas in the two states. For each county, annual mean carbon emissions from fires were calculated and summarized for the different study periods of varying lengths (i.e., 9 years for the first period of 1992–2001 and 5 years for the second period of 2001–2006).

Analyses

For the first study period (1992–2001), we assumed the average age for the NFCC was five years (i.e., half the period length between observations), but a 9-year growth period length was assumed for the areas of forest remaining forest to calculate forest carbon sequestration during the period. Similarly, we used the average age of a 3- and a 5-year growth period for the NFCC and forest remaining forest, respectively, to calculate forest sequestration during the second period (2001–2006). To calculate C loss from FNCC, we used an emission proportion factor of 0.8. This factor was based on the assumption that 80% of the nonsoil forest C (including live tree, standing dead, understory, down dead wood, and forest floor) would be lost during conversion to nonforest (Smith and Heath 2011).

Our estimates did not include soil carbon because of data limitations and because we expect little change in soil carbon. Soil carbon dynamics are usually affected most by agriculture-related land conversions. In our study area the dominant land cover type was rangeland, which would be expected to show relatively small soil carbon changes (USDA 2009). We also assumed zero change in soil carbon due to harvest on forest remaining forest because the scientific literature (for example, Nave et al. [2010]) continues to demonstrate that mineral soil carbon does not change significantly due to harvest.

Net carbon change during the study period was calculated as

$$C_{Net} = C_{FRF} + C_{NFCC} + C_{FNCC} + C_{Harvest} + C_{Fire} \quad (2)$$

Table 1. Forest cover changes (km^2) and net forest ecosystem carbon change (1,000 tonnes) associated with forest-related land cover changes from 1992 to 2006 in Oklahoma (OK) and Texas (TX), and other disturbances. Negative values indicate carbon sources whereas positive values represent carbon sinks.

	Forestland cover (km^2)			Carbon dynamics by land cover, harvest, and fire					
State	NFCC[1]	FNCC[2]	FRF	NFCC	FNCC	FRF [3]	Harv.[4]	Fire	Net[5]
				1992–2001					
OK	661	1,924	38,337	289	−10,023	26,176	−11,309	−1,241	3,892 (0.4)
TX	2,503	7,593	70,034	1,748	−35,018	67,482	−56,040	−472	−22,299 (−2.5)
Total	3,164	9,517	108,371	2,037	−45,040	93,658	−67,349	−1,713	−18,407 (−2.0)
				2001–2006					
OK	332	756	38,273	45	−3,958	14,529	−6,283	−690	3,644 (0.7)
TX	888	2,872	69,262	302	−14,843	34,740	−23,040	−262	−3,103 (−0.6)
Total	1,220	3,628	107,535	347	−18,800	49,269	−29,323	−952	541 (0.1)
				1992–2006					
OK	994	2,680	38,273	334	−13,980	40,705	−17,592	−1,931	7,536 (0.5)
TX	3,391	10,465	69,262	2,050	−49,860	102,222	−79,080	−735	−25,403 (−1.8)
Total	4,384	13,146	107,535	2,384	−63,841	142,927	−96,672	−2,665	−17,867 (−1.3)

[1] NFCC = nonforest to forest cover change.
[2] FNCC = forest to nonforest cover change.
[3] FRF = forest remaining forest.
[4] Net carbon removals after deducting amount of harvested carbon stored in landfill and wood products.
[5] Net carbon change for the corresponding periods = ($C_{Frf} + C_{NFCC} + C_{FNCC} + C_{Harvest} + C_{fire}$). Numbers in parentheses are the average for the period, in units of Tg C yr^{-1}.

Table 2. Comparison of forest area and forest ecosystem carbon emissions (percentage) from major disturbance types from 1992 to 2006 in Oklahoma (OK) and Texas (TX).

State	Forest area[1] (km^2)	Forest area (%)	Carbon emissions (%) FNCC[2]	Harvest	Fire
OK	39,266	35.1	21.9	18.2	72.4
TX	72,653	64.9	78.1	81.8	27.6
Total	111,919	100	100	100	100

[1] Based on 2006 estimates from the National Land Cover Dataset change map.
[2] FNCC = forest to nonforest cover change.

where *NFCC* = nonforest to forest cover changes, *FRF* = forest remaining forest, *FNCC* = forest to nonforest cover changes, and $C_{Harvest}$ = emissions from harvest, such that the harvested carbon stored in the longer term in wood products in use and landfills was estimated to remain stored. By definition for this study, emissions have negative values, so FNCC, harvest, and fire estimates are negative. Consequently, adding these values in this equation results in emissions being subtracted from the carbon sinks that are reported as positive values.

Results

Forest Carbon Dynamics by State

Based on RS-based land cover change maps, there was a total of 111,900 km^2 forestland as of 2006, of which 44% (or 49,700 km^2) was in the eastern portion of the region. Forest ecosystems in the entire area were estimated to be a carbon source during the period of 1992 to 2006 at an annual rate of −1.3 Tg C yr^{-1} (Table 1). However, Oklahoma was a carbon sink with annual sequestration of 0.5 Tg C yr^{-1} while Texas was identified as a carbon source with emissions of −1.8 Tg C yr^{-1}. Carbon emissions attributed to harvest accounted for 59% of all carbon emissions caused by major disturbances, followed by FNCC (39%), and fire (2%) (Table 1). Relatively high emissions from FNCC (78.1% of the total FNCC emissions) and harvest (81.8% of total harvest-related emissions), and relatively low emissions from fire (27.6% of fire-related emissions) were observed in Texas. In terms of forest extent, Texas contains 65% of the total forest area in the two states (Table 2), based on the approach used in this study.

An increasing annual forest carbon sequestration rate was observed in Oklahoma, from 0.4 Tg C yr^{-1} for the period of 1992 to 2001 to 0.7 Tg C yr^{-1} for the period of 2001 to 2006. This increase was caused by a decrease in annual net FNCC rate (calculated as the gross FNCC rate minus the NFCC rate then divided by forest area in the starting year of a given period and the number of years for the period) between the two periods, from 0.35% per year on average in the first period to 0.22% in the second period (Table 1). A similar trend was also detected in Texas, where the annual net FNCC rate was reduced to 0.55% per year in the second period from 0.74% per year in the first period. Furthermore, the annual carbon emission rate resulting from harvest was 26% lower in the period of 2001–2006 than that in the first period in Texas. As a result, forest ecosystems in Texas became a weaker carbon source in the second period with average annual emissions of −0.6 Tg C yr^{-1}, compared to −2.5 Tg C yr^{-1} in the first period. The two states as a whole functioned as a carbon sink at a rate of 0.1 Tg C yr^{-1} for the period of 2001 to 2006, compared to a carbon source of −2.0 Tg C yr^{-1} for the period of 1992 to 2001 because of reductions in annual rates of both FNCC conversion and harvest.

Forest Carbon Dynamics by Ecological Region

In general, forest ecosystem carbon dynamics during the period of 1992–2006 differed across the East to the C&W, and these differences are associated with general distribution patterns in forest type (Figure 1). Whereas the East portions of both states were identified as carbon sources, the C&W portions as a whole were carbon sinks, 17 Tg C for C&W Oklahoma and 40 Tg C for C&W Texas (Table 3). The sink in C&W Oklahoma was greater in absolute value than the source from disturbances in the eastern portion (17 Tg C versus −9 Tg C) so that the state functioned as a carbon sink overall. However, even though the sink in C&W Texas (40 Tg C) was larger than that of Oklahoma during the same period, the sink was not large enough to offset the even larger source in the East portion of the state (-65 Tg C), so Texas as a whole was a carbon source over the entire period (Table 3). Similar patterns were found in each of the two subperiods although allocations between sinks and sources varied in quantity.

Our analysis showed much smaller forest carbon sequestration in East Oklahoma for the period of 1992 to 2001 (12 Tg C), compared

Table 3. Summary of net forest ecosystem carbon change (1,000 tonnes) from 1992 to 2006 in OK and TX by ecological division (East versus C&W; see Figure 1). Negative values indicate carbon sources and positive values represent carbon sinks.

State	Division	Carbon dynamics by land cover change, harvest, and fire					
		NFCC[1]	FNCC[2]	FRF[3]	Harv.[4]	Fire	Net[5]
			1992–2001				
OK	East	134	−5,254	11,606	−11,309	−642	−5,465 (−0.6)
	C&W	155	−4,769	14,571	0	−599	9,358 (1.0)
TX	East	1,399	−22,590	32,017	−56,040	−300	−45,514 (−5.1)
	C&W	349	−12,428	35,465	0	−172	23,214 (2.6)
Total	East	1,533	−27,844	43,623	−67,349	−942	−50,979 (−5.7)
	C&W	504	−17,197	50,036	0	−771	32,572 (3.6)
			2001–2006				
OK	East	38	−3,563	6,312	−6,283	−357	−3,853 (−0.8)
	C&W	7	−395	8,217	0	−333	7,496 (1.5)
TX	East	277	−12,119	15,221	−23,040	−167	−19,828 (−4.0)
	C&W	25	−2,724	19,519	0	−96	16,724 (3.3)
Total	East	315	−15,682	21,533	−29,323	−523	−23,680 (−4.7)
	C&W	32	−3,119	27,736	0	−429	24,220 (4.8)
			1992–2006				
OK	East	172	−8,817	17,918	−17,592	−998	−9,317 (−0.7)
	C&W	162	−5,164	22,788	0	−932	16,854 (1.2)
TX	East	1,676	−34,709	47,238	−79,080	−467	−65,342 (−4.7)
	C&W	374	−15,152	54,984	0	−268	39,938 (2.9)
Total	East	1,848	−43,526	65,156	−96,672	−1,465	−74,659 (−5.3)
	C&W	536	−20,316	77,772	0	−1,200	56,792 (4.1)

[1] NFCC = nonforest to forest cover change.
[2] FNCC = forest to nonforest cover change.
[3] FRF = forest remaining forest.
[4] Emissions from net carbon removals, excluding a long-term amount of harvested carbon stored in landfill and wood products (the landfill and wood products carbon remains stored).
[5] Net carbon changes for the corresponding periods = (C_{Frf} + C_{NFCC} + C_{FNCC} + $C_{Harvest}$ + C_{fire}). Numbers in parentheses are the average for the period, in units of Tg C yr^{-1}.

to the estimate of 37 Tg C reported in our earlier national-scale study (Zheng et al. 2011). The same trend was also found in East Texas where forest carbon sequestration was estimated as 33 Tg C for the study period, compared to 48 Tg C in the national-scale study. These differences were primarily caused by differences in updated forest growth rates used in the two studies, which we discuss further below. Another substantial difference in this study, as compared to the previous national-scale study, was due to the effect of forest fire on forest ecosystem carbon change in East Oklahoma. Results from this study indicated that carbon emissions from forest fires were −0.6 Tg C (or 78% lower) in East Oklahoma for the period of 1992–2001 (Table 3), compared to −2.9 Tg C reported in Zheng et al. (2011).

Spatial Pattern of County-Level Annual Net Forest Ecosystem Carbon Change

Spatially, 50% of all counties in East Oklahoma were carbon sources, whereas only 16% of counties in C&W Oklahoma were carbon sources (excluding a few counties that have no data). The corresponding percentages serving as carbon sources were 95% of counties in eastern Texas and 21% of counties in C&W Texas (Figure 2). While most counties identified as carbon sinks were concentrated in the central portion of the states, the majority of these counties (83%) was classified in the low-sink category during the study period (Figure 2). Most counties identified as moderate or strong carbon sources during the period were located in the eastern portions of both states. In part, this was caused by the fact that about 44% of all forestlands were located in the eastern portion of the states, and the relatively high carbon stocking of the eastern forests contributed to a greater impact from disturbance on a per area basis. Moreover, much higher forest-related disturbance rates occurred in the East. The net FNCC rate in the eastern portion of the states was 55% higher than that in the C&W (9.0% versus 5.8% during the period of 1992 to 2006). Most carbon removals from commercial forest harvests occurred in the East portions of both states (Li et al. 2010). Carbon emissions from fires were also slightly higher (55%) in the East (Table 3).

Discussion

Annual net forest growth estimates used in this study were lower, on average, than those available for the previous broader scale study (Zheng et al. 2011). The growth estimates for loblolly pine used in this study reached their peak during age 10–15 years whereas the regional data used in the previous study featured a general plateau between about 5–25 years old, after which they began to decline (Figure 3). For oak/hickory, net growth reached its peak value at ages of 20–25 years based on the data used in this study, whereas regional net growth used by Zheng et al. (2011) peaked at 35–40 years old. Average growth for both forest types in Texas and Oklahoma were much slower than those from the broader regional data. In loblolly pine, the four 5-year age classes ranging from 5 to 25 years old had higher growth for Texas and Oklahoma, but these age classes accounted for only about 10% of the total area of forest remaining forest in the two states. The effects of lower growth at more advanced stand ages is a major factor for reducing the carbon estimates of this region compared to Zheng et al. (2011).

A substantial difference in estimated carbon emissions from forest fires between this study and a previous study (Zheng et al. 2011) was found for Oklahoma. We estimated −1.2 Tg C was released from fires for 1992 to 2001 in this study (Table 1), compared to −2.9 Tg C from the previous study, a 57% reduction. This reduction is primarily due to differences in estimated burn areas from updated data. For example, the mean annual burn area in this study was about 12,296 ha calculated from 14-year fire data (1992 to 2006), 63% lower than the estimated annual burn area of 33,156 ha in the previous study resulting from a three-year average

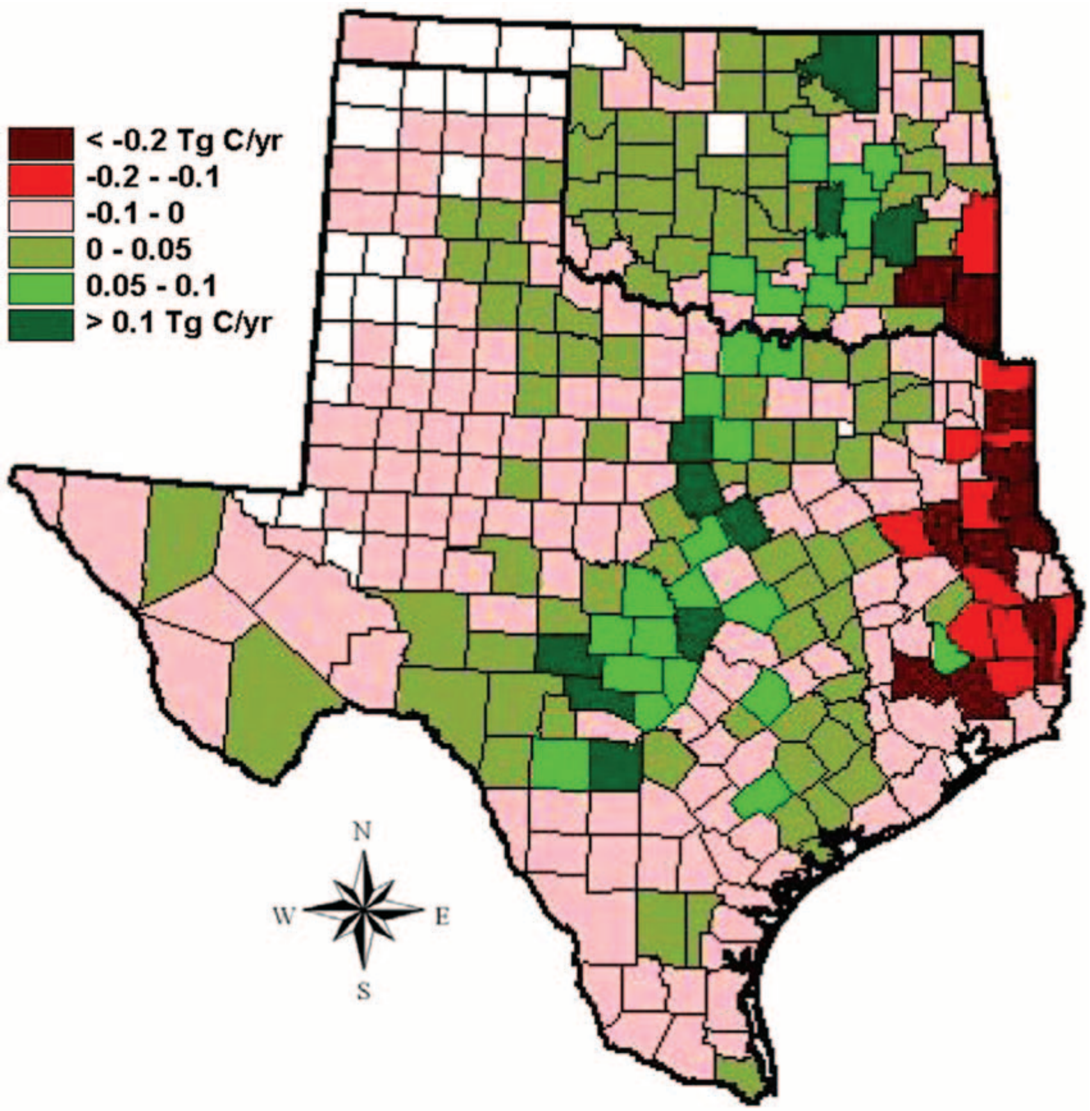

Figure 2. Spatial distribution of annual net forest ecosystem carbon change for 1992 to 2006 at county level in Oklahoma and Texas. Negative values indicate carbon sources and positive values are carbon sinks. Counties in white have no estimates due to lack of data. Dark lines are state boundaries and lighter lines are county boundaries.

(2004–2006) due to data limitations (Zheng et al. 2011). The differences obtained here show that mean annual burn areas obtained from relatively long-term fire data are more suitable for net forest ecosystem carbon estimation over long periods at broad scales. Short-term averages can potentially overstate or understate the impact of fire on carbon dynamics. Partitioning carbon emissions in different disturbance types may also vary depending on availability of data sources and methodologies used in the analyses. For example, this study indicated that carbon emission through wildfires accounted for 2% of total emission from all three disturbance types in the study area with unknown estimation error. Our method is comparable with a previous report that the contribution of carbon emission through wildfires was generally very low (0.6%) among the three general disturbance types in the southern US states (including Oklahoma and Texas), compared to those identified for the other regions (Zheng et al. 2011).

The differences in growth rates resulted in a notable shift in net forest ecosystem carbon change for East Oklahoma for the period of 1992 to 2001, from a sink of 16 Tg C reported in a previous study (Zheng et al. 2011) to a source of -5 Tg C in this study (Table 3). However, the whole state of Oklahoma was still a carbon sink for the period of 1992 to 2001 because of net carbon gains from forestlands in the C&W portions, which was not included in the previous study due to data limitations.

In East Texas, net forest ecosystem carbon changes were similar, -3.9 Tg C yr^{-1} (Zheng et al. 2011) as compared to -4.0 Tg C yr^{-1} (Table 3) during the period of 1992 to 2001 as total growth increased for the state but FNCC emissions increased even more. Carbon emission for the entire state, however, were reduced to -2.5 Tg C yr^{-1} due to offset of net carbon sequestration in the C&W portions of the state that had limited disturbances. The differences suggest that using localized forest growth estimates, and a more complete accounting of low-productivity but broadly distributed forests, are important when using aggregated growth estimates at a broader scale.

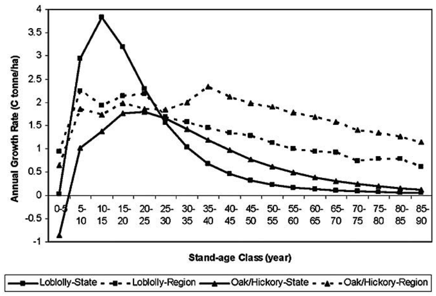

Figure 3. Comparisons between annual net forest growth rates obtained for this study and the rates aggregated at a broader geographic region (i.e., in the south-central region of the United States, Smith et al. 2006) in a previous study (Zheng et al. 2011) for two representative common forest types in the two states: loblolly pine and oak/hickory. Negative values indicate carbon emissions whereas positive numbers designate carbon gains.

This analysis also included carbon in products and landfills. We estimate that 12% of total harvested carbon in the two states remained in these pools after 100 years, a period of time that is used to indicate permanent storage. Results from this study could underestimate the overall C emission in the C&W portion of Oklahoma and Texas because of the assumption there were no C removals from harvesting, possibly due to limited data from the FIA dataset. However, based on the data that less than 2% of harvests in Texas are outside of the East, C emissions in the C&W portion of Oklahoma and Texas were underestimated by approximately 0.12 Tg C yr^{-1}. Some of the harvested material may also generate significant offsets of fossil fuel use when substituted for less energy-intensive materials or by direct substitution as an energy source. However, the carbon- and climate-neutrality of such removals remains controversial (Malmsheimer et al. 2011, Schulze et al. 2012) and a full accounting of impacts would require consideration of both greenhouse gases and biophysical impacts over extended time scales (Anderson 2010, Bright et al. 2012). Such an analysis is beyond the scope of this study.

Uncertainty

The greatest uncertainty in these carbon estimates comes from identification of forest area and change in the region. Differences in forest area estimates from RS- and the changing FIA-based observations were mainly caused by differences in definition of forestland and in the approach used to estimate areas (Zheng et al. 2011). Additional differences are associated with the decision to include or exclude areas mapped as woody wetland and shrub/scrub using RS-based data in the definition of forest. Our RS-based observations suggested forest areas (deciduous, evergreen, and mixed) were about 39,000 km^2 and 73,000 km^2 (NFCC plus forest remaining forest), respectively, in Oklahoma and Texas in 2006 (Table 1). Smith et al. (2009) reported that FIA used RS-based area estimates in west Texas and west Oklahoma for their forest statistics at that time and all previous reports, and the forest area results for the states for 2006 are similar to ours: about 31,160 km^2 and 70,000 km^2, respectively, in Oklahoma and Texas. However, based on the initial survey results of the traditional FIA-based data approach, Cooper (2013) and Bentley (2012) report forest areas of about 51,200 km^2 (of which 20,200 km^2 is categorized as "other forestland") and 244,600 km^2 (of which 193,800 km^2 is categorized as "other forestland") in Oklahoma and Texas, respectively. Including woody wetlands and shrub/scrub classes in the NLCD would increase our RS-based estimates of area, but we are using the classes traditionally used to represent forestland. In this study, we are focused on carbon change over time, so unfortunately having only one FIA survey or a partial survey available for the entire state does not provide area change estimates. Furthermore, these data were taken for years more recent than 2006, and our analysis period is before these years. Therefore, we use the RS-based observations that provide area change estimates, and are similar to the estimates published by FIA, such as in Smith et al. (2009), which are also based on remote sensing.

In the future, the sudden appearance of such a large area of "other forestland" in these states is going to need to be dealt with in a carbon or greenhouse gas inventory. One way to deal with this is to calculate the inventory for the entire land base, including rangelands and wetlands, for example, in addition to forestland. This would provide a comprehensive land-based estimate and ensure that definitional changes do not result in large carbon changes that could be easily misinterpreted.

If forest carbon emissions from various disturbance sources are kept constant but much larger forest areas are used in carbon calculations, the conclusions regarding forest carbon dynamics during the study periods may differ. Additional mapping errors also existed in the NLCD data. For example, overall accuracy of the 1992 NLCD map at the Anderson level I was 74% with a standard error of 2% for the south-central region, about 9% lower than the mean accuracy of other five regions in the western United States (Wickham et al. 2004). Temporary changes in land cover caused by unexpected natural disasters, such as hurricanes in the area (Thompson et al. 2011), could also affect accuracy of forest ecosystem carbon estimates using our approach.

Another source of uncertainty is the possible designation of harvested areas as FNCC. We investigated the possible impacts of double-counting by conducting a basic sensitivity analysis, assuming that half of the sequestration and half of the emissions from NFCC and FNCC were actually from areas that were harvested and, therefore, continue to be FRF. Results indicated that Oklahoma would be a carbon sink of 1.0 Tg C yr^{-1}, and Texas would be a small carbon source of -0.1 Tg C yr^{-1}, as compared to 0.5 Tg C yr^{-1} and -1.8 Tg C yr^{-1}, respectively. These are major differences, and more work is needed to ensure harvests are not double-counted.

Fire data are also a source of uncertainty. Although some fires smaller than the thresholds were excluded in the MTBS maps, effects of such omission are very limited. For example, the MTBS maps used in this study, generally speaking, covered 98% of all fires in terms of burn area for the period of 1992–2006. However, substantial variations in burn area from year to year (or period to period) exist and can affect estimation of carbon emissions from fires. Thus, fire data (i.e., burn area and severity) obtained from longer periods are relatively more reliable. Unfortunately, to our knowledge, there has been no uncertainty analysis of the fire data (Brad Quayle, USDA Forest Service, pers. comm., March 25, 2013).

Finally, forest growth rates contain uncertainty; they could be affected by several factors such as forest type, forest age structure, the data availability across the study area, and the method used for the rate calculation. Our data suggested that the overall mean forest growth rate for the region was 0.94 t C $ha^{-1}yr^{-1}$ within a 95% confidence interval ranging from 0.78 t C $ha^{-1}yr^{-1}$ to 2.73 t C $ha^{-1}yr^{-1}$ (Table 4). The mean growth rates by state were 0.76 and 1.04 t C $ha^{-1}yr^{-1}$, respectively, for Oklahoma and Texas and were 0.92 and 0.98 t C $ha^{-1}yr^{-1}$ for the C&W and East regions of the

Table 4. Mean forest growth (t C/ha/yr) with 95% confidence interval in OK and TX and ecoregion (East versus C&W) between 1992 and 2006, based on county-level weighting of forest area and forest group type.

	East			C&W			All		
State	Low-95	Mean	High-95	Low-95	Mean	High-95	Low-95	Mean	High-95
OK	−1.185	0.624	2.520	−0.936	0.913	2.804	−1.070	0.759	2.652
TX	−0.539	1.244	3.108	−0.664	0.915	2.549	−0.616	1.044	2.769
Total	−0.814	0.98	2.858	−0.746	0.915	2.624	−0.776	0.944	2.728

two states, respectively. Forest growth for the counties in C&W regions of the two states were developed based on data availability that may not include the same number of years due to different inventory schedules for different states. Data on utilization rates and logging residue decay rates are limited.

Despite these limitations, this study provides an approach that is consistent over the time period of interest to estimate forest ecosystems carbon dynamics associated with forest growth and major disturbances. The time period of interest is important because it is needed for reporting to an international convention that the United States has indeed ratified.

Conclusions

The forests of the two-state area were estimated as a small carbon source at an annual rate of −1.3 Tg C, which is about 1% (in absolute value) of annual net forest carbon changes in the 48 states (Zheng et al. 2011). Carbon change estimated from this study using updated disturbance and growth data covering the entire states of Oklahoma and Texas confirm that forest ecosystems in Oklahoma were a carbon sink for the period from 1992 to 2001, whereas Texas was a carbon source, with an overall rate of −2.0 Tg C yr^{-1} for the study area. This is consistent with the estimates reported from a previous study using forest data in eastern portion of the two states alone (Zheng et al. 2011). In contrast, the two states were a small annual sink in the early 2000s (0.1 Tg C yr^{-1} as compared to −2.0 Tg C yr^{-1}) due to reduced FNCC and harvesting.

Localized forest growth data can differ substantially from those generated at a coarser resolution but should better represent reality. Our results suggest that using localized forest growth estimates and a more complete accounting of low-productivity but broadly distributed forests are important for a more accurate estimate of net carbon change than can be obtained when using aggregated growth estimates at a broader scale and low-productivity forests are omitted. Annual data on forest fires for a longer period also notably affected results. A more careful examination of harvesting is needed to ensure emissions from areas of cover change are not double-counted as harvest emissions.

Finally, in the future, analyses need to include recent estimates of large areas of forest categorized as "other forestland" and determine their effects on land-based carbon changes. One approach would be to use consecutive FIA surveys and take the difference, once a second survey is conducted in C&W Texas and Oklahoma over the coming years and the data become available. Another option would be to include the entire land base in the analyses. This would also provide estimates for rangeland, wetlands, and other lands and would help provide information to support a broad perspective about the shifts in land area in the various land cover categories.

Literature Cited

ANDERSON, J.R., E.E. HARDY, J.T. ROACH, AND R.E. WITMER. 1976. A land use and land cover classification system for use with remote sensing data. USGS Professional Pap. 964, Reston, VA. 28 p.

ANDERSON, R.G., J.G. CANADELL, J.T. RANDERSON, R.B. JACKSON, B.A. HUNGATE, D.D. BALDOCCHI, ET AL. 2010. Biophysical considerations in forestry for climate protection. *Frontiers Ecol. Environ.* 9:174–182.

BENTLEY, J.W. 2012. *Texas, 2010 forest inventory and analysis factsheet. e-Science Update SRS–045.* 5 p. Available online at www.srs.fs.usda.gov/pubs/su/su_srs045.pdf; last assessed Aug. 10, 2013.

BRIGHT, R., F. CHERUBINI, R. ASTRUP, D.N. BIRD, A. COWIE, M. DUCEY, ET AL. 2012. A comment to "Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral": Important insights beyond greenhouse gas accounting. *Glob. Change Biol. Bioenerg.* 4:617–619.

CHEN, X., S. LIU, Z. ZHU, J. VOGELMANN, Z. LI, AND D. OHLEN. 2011. Estimating aboveground forest biomass carbon and fire consumption in the US Utah High Plateaus using data from the Forest Inventory and Analysis Program, Landsat, and LANDFIRE. *Ecol. Indic.* 11:140–148.

COOPER, J.A. 2013. *Oklahoma, 2011—Forest inventory and analysis factsheet.* e-Science Update SRS-SU-066. 5 p. Available online at www.srs.fs.usda.gov/pubs/43623; last assessed Aug. 10, 2013.

COULSTON, J.W., S.N. OSWALT, A.B. CARRAWAY, AND W.B. SMITH. 2010. Assessing forestland area based on canopy cover in a semi-arid region: A case study. *For.* 83:143–151.

DRUMMOND, M.A., AND T.R. LOVELAND. 2010. Land-use pressure and a transition to forest-cover loss in the eastern United States. *BioSci.* 60:286–298.

EIDENSHINK, J., B. SCHWIND, K. BREWER, Z. ZHU, B. QUAYLE, AND S. HOWARD. 2007. A project for monitoring trends in burn severity. *Fire Ecol.* 3:3–21.

FRY, J.A., M.J. COAN, C.G. HOMER, D.K. MEYER, AND J.D. WICKHAM. 2009. *Completion of the National Land Cover Database (NLCD) 1992–2001 land cover change retrofit product.* USGS, Open-File Report 2008–1379, Reston, VA. 18 p.

GREEN, E.J., C.E. THOMAS, AND W.E. STRAWDERMAN. 1987. Stein-rule estimation of timber removals by county. *For. Sci.* 33:1054–1061.

HANSEN, M.C., S.V. STEHMAN, AND P.V. POTAPOV. 2010. Quantification of global gross forest cover loss. *Proc. Natl. Acad. Sci. USA* 107:8650–8655.

HANSEN, M.C., R.S. DEFRIES, J.R. TOWNSHEND, AND R. SOHLBERG. 2000. Global land cover classification at 1 km spatial resolution using a classification tree approach. *Int. J. Rem. Sens.* 21:1331–1364.

HEATH, L.S., J.E. SMITH, K.E. SKOG, D. NOWAK, AND C. WOODALL. 2011. Managed forest carbon estimates for the US Greenhouse Gas Inventory, 1990–2008. *J. For.* 109:167–173.

JOHNSON, E., G. GEISSLER, AND D. MURRAY. 2010. *The Oklahoma Forest Resource Assessment, 2010.* Oklahoma Forestry Division, Oklahoma Department of Agriculture, Food and Forestry. 162 p. Available online at www.forestry.ok.gov/2010-assessment-final-documents; last accessed Mar. 31, 2013.

LEI, Y.C., AND S.Y. ZHANG. 2004. Features and partial derivatives of Bertalanffy–Richards growth model in forestry. *Nonlinear Anal. Model. Contr.* 9:65–73.

LI, Y., B. CARRAWAY, AND C. VANDERSCHAAF. 2010. *Harvest trends, 2009.* Texas Forest Service. 21 p. Available online at http://texasforestservice.tamu.edu/uploadedFiles/Sustainable/econdev/Publications/Reports/HarvestTrends2009.pdf; last accessed Mar. 31, 2013.

MALMSHEIMER, R.W., J.L. BOWYER, J.S. FRIED, E. GEE, R.K. IZLAR, R.A. MINER, ET AL. 2011. Managing forests because carbon matters: Integrating energy, products, and land management policy. *J. For.* 7S:S7–S51.

MILLS, J.R., AND J.C. KINCAID. 1992. The aggregate timberland assessment system—ATLAS: A comprehensive timber projection model. USDA For. Serv., Gen. Tech. Rep. PNW-GTR-281, Portland, OR. 168 p.

MINER, R. 2006. The 100-year method for forecasting carbon sequestration in forest products in use. *Mitig. Adapt. Strateg. Glob. Change.* Available online at www.springerlink.com/content/2l672741l7366751/; last accessed Mar. 31, 2013.

NAVE, L.E., E.D. VANCE, C.W. SWANSTON, AND P.S. CURTIS. 2010. Harvest impacts on soil carbon storage in temperate forests. *For. Ecol. Manage.* 259:857–866.

PACALA, S.W., G.C. HURTT, D. BAKER, P. PEYLIN, R.A. HOUGHTON, R.A. BIRDSEY, ET AL. 2001. Consistent land- and atmosphere-based US carbon sink estimates. *Science* 292:2316–2320.

Potter, C., S. Klooster, A. Huete, and V. Genovese. 2007. Terrestrial carbon sinks for the United States predicted from MODIS satellite data and ecosystem modeling. *Earth Interact.* 11:1–21.

Reams, G.A., C.K. Brewer, and R.W. Guldin. 2010. Remote sensing alone is insufficient for quantifying changes in forest cover. *Proc. Natl. Acad. Sci. USA* 107:E145.

Schulze, E.-D., C. Körner, B. Law, H. Haberl, and S. Luyssaert. 2012. Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral. *Glob. Change Biol. Bioenerg.* 4:611–616.

Smith, J.E., and L.S. Heath. 2011. Carbon stocks and stock changes in US forests, and Appendix C. P. C-1–C-7 in *USDA US Agriculture and Forestry Greenhouse Gas Inventory: 1990–2008.* Tech. Bull. 1930. 162 p. Available online at www.usda.gov/oce/climate_change/AFGG_Inventory/USDA_GHG_Inv_1990–2008_June2011.pdf; last accessed Aug. 10, 2013.

Smith, J.E., L.S. Heath, and M.C. Nichols. 2010. US forest carbon calculation tool: Forest-land carbon stocks and net annual stock change: Revised for use with FIADB4.0. USDA For. Serv., Gen. Tech. Rep. NRS-GTR-13, Newtown Square, PA. 34 p.

Smith, J.E., L.S. Heath, K.E. Skog, and R.A. Birdsey. 2006. Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States. USDA For. Serv., Gen. Tech. Rep. NE-GTR-343, Newtown Square, PA. 216 p.

Smith, W.B., P.D. Miles, C.H. Perry, and S.A. Pugh. 2009. Forest resources of the United States, 2007. USDA For. Serv., Gen. Tech. Rep. WO-GTR-78, Washington, DC. 336 p.

Thompson, B.K., F.J. Escobedo, C.L. Staudhammer, C.J. Matyas, and Y. Qiu. 2011. Modeling hurricane-caused urban forest debris in Houston, Texas. *Landscape Urban Plan.* 101:286–297.

Turner, D.P., G.J. Koerper, M.E. Harmon, and J.J. Lee. 1995. A carbon budget for forests of the conterminous United States. *Ecol. Appl.* 5:421–436.

USDA. 2009. *Summary report: 2007.* National Resources Inventory, Natural Resources Conservation Service, Washington, DC, and Center for Survey Statistics and Methodology. Available online at www.nrcs.usda.gov/technical/NRI/2007/2007_NRI_Summary.pdf; last accessed Aug. 1, 2012.

USDA Forest Service. 2011. *FIA DataMart. FIADB 4.0.* Available online at http://apps.fs.fed.us/fiadb-downloads/datamart.html; last accessed Aug. 17, 2011.

USDA Forest Service. 2012. *Forest timber product output (TPO) reports.* Available online at http://srsfia2.fs.fed.us/php/tpo_2009/tpo_rpa_int1.php; last accessed Jan. 20, 2012.

Wickham, J.D., S.V. Stehman, J.H. Smith, and L. Yang. 2004. Thematic accuracy of the 1992 National Land-Cover Data for the western United States. *Rem. Sens. Environ.* 91:452–468.

Woodbury, P.B., J.E. Smith, and L.S. Heath. 2007. Carbon sequestration in the US forest sector from 1990 to 2010. *For. Ecol. Manage.* 241:14–27.

Woudenberg, S.W., B.L. Conkling, B.M. O'Connell, E.B. LaPoint, J.A. Turner, and K.L. Waddell. 2010. The Forest Inventory and Analysis database: Database description and users manual version 4.0 for Phase 2. USDA For. Serv., Gen. Tech. Rep. RMRS-GTR-245, Fort Collins, CO. 336 p.

Xian, G., C. Homer, and J. Fry. 2009. Updating the 2001 National Land Cover Database land cover classification to 2006 by using Landsat imagery change detection methods. *Rem. Sens. Environ.* 113:1133–1147.

Zeide, B. 1993. Analysis of growth equations. *For. Sci.* 39:549–616.

Zhang, F., J.M. Chen, Y. Pan, R.A. Birdsey, S. Shen, W. Ju, and L. He. 2012. Attributing carbon changes in conterminous US forests to disturbance and non-disturbance factors from 1901 to 2010. *J. Geophys. Res.* 117:G02021, doi:10.1029/2011JG001930.

Zheng, D., L. Heath, M. Ducey, and B. Quayle. 2013. Assessing double counting of forest carbon emissions between deforestation and wildfires: A case study in the United States, 1992–2006. *Ecosystems* 16:310–322.

Zheng, D., L.S. Heath, M.J. Ducey, and J.E. Smith. 2011. Carbon changes in conterminous US forests associated with growth and major disturbances: 1992–2001. *Environ. Res. Lett.* 6:014012.

Zheng, D., J. Rademacher, J. Chen, T. Crow, M. Bresee, J. LeMoine, and S.-R. Ryu. 2004. Estimating aboveground biomass using Landsat 7 ETM+ data across a managed landscape in northern Wisconsin, USA. *Rem. Sens. Environ.* 93:402–411.

Zheng, D., S.D. Prince, and R. Wright. 2003. Terrestrial net primary production estimates for 0.5^0 grid cells from field observations—A contribution to global biogeochemical modelling. *Glob. Change Biol.* 9:46–64.