ECOSYSTEMS CrossMark

A Tale of Two Forest Carbon Assessments in the Eastern United States: Forest Use Versus Cover as a Metric of Change

C. W. Woodall,[1]* B. F. Walters,[1] M. B. Russell,[2] J. W. Coulston,[3] G. M. Domke,[1] A. W. D'Amato,[4] and P. A. Sowers[1]

[1]Northern Research Station, Forest Inventory and Analysis Program, USDA Forest Service, St. Paul, Minnesota, USA; [2]Department of Forest Resources, University of Minnesota, St. Paul, Minnesota, USA; [3]Southern Research Station, Forest Inventory and Analysis Program, USDA Forest Service, Blacksburg, Virginia, USA; [4]Rubenstein School of Environment and Natural Resources, University of Vermont, Burlington, Vermont, USA

ABSTRACT

The dynamics of land-use practices (for example, forest versus settlements) is often a major driver of changes in terrestrial carbon (C). As the management and conservation of forest land uses are considered a means of reducing future atmospheric CO_2 concentrations, the monitoring of forest C stocks and stock change by categories of land-use change (for example, croplands converted to forest) is often a requirement of C monitoring protocols such as those espoused by the Intergovernmental Panel on Climate Change (that is, Good Practice Guidance and Guidelines). The identification of land use is often along a spectrum ranging from direct observation (for example, interpretation of owner intent via field visits) to interpretation of remotely sensed imagery (for example, land cover mapping) or some combination thereof. Given the potential for substantial differences across this spectrum of monitoring techniques, a region-wide, repeated forest inventory across the eastern U.S. was used to evaluate relationships between forest land-use change (derived from a forest inventory) and forest cover change (derived from Landsat modeling) in the context of forest C monitoring strategies. It was found that the correlation between forest land-use change and cover change was minimal (< 0.08), with an increase in forest land use but a net decrease in forest cover being the most frequent observation. Cover assessments may be more sensitive to active forest management and/or conversion activities that can lead to confounded conclusions regarding the forest C sink (for example, decreasing forest cover but increasing C stocks in industrial timberlands). In contrast, the categorical nature of direct land-use field observations reduces their sensitivity to forest management activities (for example, clearcutting versus thinning) and recent disturbance events (for example, floods or wildfire) that may obscure interpretation of C dynamics over short time steps. While using direct land-use observations or cover mapping in forest C assessments, they should not be considered interchangeable as both approaches possess idiosyncratic qualities that should be con-

sidered when developing conclusions regarding forest C attributes and dynamics across large scales.

INTRODUCTION

As humans have either directly (via development) or indirectly (via climate altering greenhouse gas emissions) affected the entire surface of our planet (Ellis and others 2010), the monitoring of how land is used has been identified as critical to the sustainability of ecosystems and their associated processes such as terrestrial carbon (C) cycles (Brown and others 2013). Land use is often defined as the purpose for which land is used by humans (Brown and others 2013). Although land use can be discretely categorized (for example, croplands, forests, or settlements), it is often the concept of ''human purpose'' that complicates any land-use assessment. First, the true intent of any land owner may never be known even if the current condition of the land is observed (for example, regenerating forest that an owner intends to be cropland). Second, as direct interpretations and discussions with individual land owners across large scales are often not practical, land cover assessments are often used as a surrogate for land use. For example, coarse resolution remotely sensed imagery consistently collected across an entire nation can delineate vegetative cover types (for example, forest versus grasslands), which in turn enables the monitoring of land use in the absence of information regarding human intent. The juxtaposition of directly observed land use in the context of owner intent versus vegetative cover mapping can lead to divergent conclusions regarding true land use. For example, if a forest has complete canopy cover a land cover assessment will estimate a forest land use. If a forest has been recently clear cut and replanted it may be considered a nonforest land use (for example, cropland) by a land cover assessment although a land-use survey may estimate the parcel as remaining a forest. Both land use and cover monitoring approaches possess their own sources of error and latency (for example, field remeasurement cycles and spectral interpretation, respectively). Regardless of methods for deriving land use, land use remains an important driver of numerous ecosystem processes (for example, Birdsey and others 2006; Rhemtulla and others 2009; Radeloff and others, 2012). For example, the change in land use from a forest to a settlement has profound implications regarding all ecosystem

processes being hydrological or biogeochemical. It is for this reason that the quantification of land-use change is central to assessments of the terrestrial C cycle and is fundamental to global efforts to monitor greenhouse gas emissions (for example, Intergovernmental Panel on Climate Change (IPCC) Good Practice Guidance and associated Guidelines; IPCC 2003, 2006).

Within the context of greenhouse gas accounting, assessment of land use is essential for identifying sectors of the economy (for example, agriculture, forestry, or urban development) responsible for sources or sinks of C that in turn enables policy actions aimed at reducing net C flux to the atmosphere (for example, EOP 2013). Unfortunately, when such forest C policies are explored there is often a lack of direct observations of land use at large scales (for example, national to regional). Instead, land cover assessments using remote sensing data and associated products (for example, Landsat and the National Land Cover Database) are often used as a surrogate for land use (Gibbs and others 2007) with associated modeling of forest C attributes based on activity data (for example, even-aged forest management systems) and/or default factors. A major strength of land cover approaches is the rapid assessment of widespread disturbance or management events (for example, wildfire or timber harvest) using recently acquired remotely sensed imagery to estimate postdisturbance forest C (for example, Kurz and others 2008, 2009). Although recent advances in image processing have allowed for the estimation of forest cover loss over large geographical extents using Landsat data (Hansen and others 2013), these approaches may obfuscate finer scale C dynamics associated with phase shifts within (for example, old forest to young forest) and between (for example, afforestation of agricultural lands) land-use classes (Coulston and others 2014) as previously noted. Hence, the question arises: how does the method of land-use estimation across the spectrum from field-based to remotely sensed affect subsequent conclusions regarding forest C baselines?

Given that forests are the largest terrestrial sink of C (Pan and others 2011), which can be profoundly affected by land-use change (Houghton and others 1999; Caspersen and others 2000;

Houghton 2003), estimating the effects of land-use change (that is, loss or gain of forests) and associated forest C dynamics is a prescription of the Intergovernmental Panel on Climate Change (IPCC) Good Practice Guidance and Guidelines (IPCC 2003, 2006). Such guidelines are widely followed for developing national greenhouse gas inventories (NGHGI) for annual submission to the United Nations Framework Convention on Climate Change (UNFCCC) and emulated for other reporting and monitoring initiatives (USDA 2011) across various scales (for example, Gibbs and others 2007). Often the most valuable metric of forest C dynamics is the change in forest C over time. In the U.S. NGHGI, a stock difference approach is used to estimate changes in the U.S. forest C stock where the difference in the total U.S. forest C stock is estimated at two points in time and annualized (Woodall and others 2015a). The crux of large-scale forest C monitoring often exists at this intersection of stock and land-use change (for example, UNFCCC 2013).

Nabuurs and others (2013) demonstrated the influence of land use on the dynamics of forest C across Europe, suggesting that current land-use trends may eventually lead to the loss of the European forest C sink with serious implications for European Union climate change policies. In a similar regard, land use has been identified as a strong driver of the U.S. forest C sink (Caspersen and others 2000; Woodall and others 2015b) and may be the dominant factor controlling rates of forest C accumulation in the U.S. (Rhemtulla and others 2009; Radeloff and others 2012). Given the centuries of land-use change that have occurred across the breadth of U.S. forests (Birdsey and others 2006; Nowacki and Abrams 2015) in concert with a diversity of land-use practices (for example, agricultural abandonment in the northeastern U.S., Foster 1992; conversion of agricultural lands to intensive pine plantation management in southeastern U.S., Fox and others 2007), the monitoring of land use is central to forest C management at large scales. In recognition of this knowledge gap, a UNFCCC review (UNFCCC 2013) has called upon the U.S. to refine their land-use estimation procedures to better inform forest C monitoring and policy development. Such technical improvements to C reporting and accounting may be more critical in the future due to the potential reduction in the rate of forest C sequestration (for example, Nabuurs and others 2013). Indeed, Coulston and others (2015) examined land-use change effects on forests in the southeastern U.S., finding that net forest land-use change only contributed about 6 Tg C/y

compared to net forest accumulation of about 75 Tg C/y. Furthermore, Woodall and others (2015b) found that recent land-use change constituted approximately 37% of the forest C sink in eastern U.S. forests.

The science of land-use monitoring is still evolving with the refinement of land cover mapping (Brown and others 2013) in concert with an influx of in situ information gleaned from big data sources (for example, forest inventory data, USDA 2014a). As an example, Coulston and others (2014) found that forest cover versus forest land-use assessments can differ substantially, especially across industrialized forests typical of the southeastern U.S. This is not a trivial finding as land cover assessments, which are available for much of the earth (Hansen and others 2013), are often used as a surrogate for more directly observed land-use change which follows IPCC Good Practice Guidance and Guidelines (IPCC 2003, 2006). Given the ecological and political importance of accurately documenting the global impacts of land-use change on forest C at national and regional scales (for example, U.S.' Climate Action Plan, EOP 2013), there is a need to evaluate the implications of reporting current and future forest C sink strength by divergent land-use assessments (for example, direct field observation versus vegetative reflectance modeling) across the dynamic forests of the U.S.

As forest C assessments often derive land use from a spectrum of techniques (direct field observation to modeling of vegetative reflectance) with little regard as to its influence on resulting forest C monitoring conclusions, the goal of this study was to use a systematic and repeated forest inventory to examine forest C stocks and stock change by forest land-use change and cover change across the eastern U.S. For the purpose of this study, land use will refer to the direct observation of current vegetation in the context of landowner and/or management intentions using high-resolution remotely sensed imagery combined with site visits via a forest inventory (for example, Coulston and others 2015; Woodall and others 2015b). In contrast, land cover will refer to the interpretation of spectral signals of vegetative canopies derived from Landsat data to differentiate land cover types (for example, Hansen and others 2013). Specific objectives of this study were to (1) examine differences in estimates of forest land-use change using land use versus land cover-based approaches (Hansen and others 2013), (2) evaluate correlations and sensitivity between forest C stock and stock change with land-use and cover metrics by individual forest C pool,

and (3) assess aboveground live tree and soil organic C (two largest pools) in landscapes where the largest divergences between forest land use and cover are identified in prior objectives.

METHODS

Forest Inventory Sample Design

Our study relies on the plot network and associated forest data collected by the USDA Forest Service's Forest Inventory and Analysis (FIA) program, which is the primary source for information about the extent, condition, status, and trends of forest resources across the United States (Oswalt and others 2014). The FIA program uses a nationally consistent sampling design covering all ownerships across the U.S. (Bechtold and Patterson 2005). A rotating panel repeated-measure sample design is used and is based on a global sampling design developed by White and others (1992). Each panel and the entire sample are systematic across the U.S. based on a triangular grid. The triangular grid is isotropic, and each sample location represents a hexagonal area of 2,428 ha. One permanent sampling location was randomly chosen within each hexagon. The FIA program uses a poststratified estimator (Cochran 1977) to estimate parameters of interest. Typically, remote sensing products are used for the poststratification (Westfall and others 2011). In our study, forest land use and cover were estimated at each FIA plot location using separate data sources.

Land-Use Determination

Fine-scale remotely sensed imagery (National Agriculture Imagery Program—NAIP; NAIP 2015) was used to assign the land use at each sample location at two points in time with a nominal spatial resolution (raster cell size) of 1 m^2 (Figure 1). Prior to field measurement of each year's collection of annual plots due for measurement (that is, panel), each sample location in the panel is photo interpreted manually by a forester to determine land use (Table 1). Forest was defined as land at least 37 meters wide and at least 0.4 hectares in size with at least 10% cover (or equivalent stocking) by live trees including land that formerly had live tree cover and that will be naturally or artificially regenerated (Oswalt and others 2014). Trees were defined as woody plants having a more or less erect perennial stem(s) capable of achieving at least 7.6 cm in diameter at breast height, or 12.7 cm diameter at root collar, and a height of 5 m at maturity in situ. Those sample locations determined to be in forest land use, potentially forest land use, or were forest land use at the previous measurement are field verified to finalize the land-use designation. Forest measurements are collected at sample locations designated as forest land use.

Forest Cover Determination

At each FIA plot, forest cover change was calculated based on Hansen and others' (2013) land cover change product (Figure 1). Polygons of FIA plots based on exact coordinates were intersected with Hansen's raster data to derive the fraction of plot area with forest cover increase/decrease that roughly aligns with the measurement interval of FIA's plot network. A net forest cover change was estimated from the addition of the plot area fractions of cover increase/decrease although more expanded analyses could also explore the gross changes in cover gains and losses. The range in acquisition years (2000–2012) for the Landsat data used in Hansen and others' (2013) product roughly aligns with the date range (2002–2012) of forest inventories conducted for the determination of land-use change given the 5-year remeasurement period of the forest inventory in eastern U.S. states (for example, Oswalt and others 2014).

Forest Measurements

Only forested land (use) was measured for site/vegetative attributes in the field component of the inventory. If a plot was a nonforest land use at either measurement time, its forest C stocks (for example, live tree biomass or dead wood) were assumed to be zero. This is a challenge when applying the stock difference method (for example, consistent C inventories across various land uses), which can lead to erroneous conclusions for stocks associated with pools such as soil organic C. There is no immediate emission of C when soil organic C transitions from one land use to another. Given the potential effect of these assumptions on study results, changes in soil organic C were delineated from other forest C pools in subsequent analyses to explore potential effects on conclusions regarding C dynamics. For forested inventory plots, tree and site attributes were measured for plots established with a sampling intensity of approximately one plot per 2,428 ha. Forest inventory plots established in forested conditions consisted of four, 7.32-m fixed-radius subplots spaced 36.6 m apart in a triangular arrangement with one subplot in the center (USDA 2014a, b, c). All trees (live and standing dead) with a diameter at breast height of at least 12.7 cm were inventoried on forested subplots. A standing dead
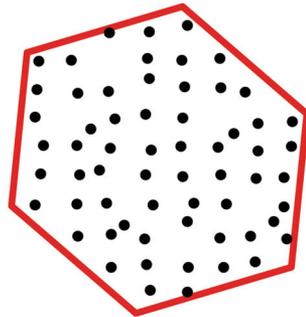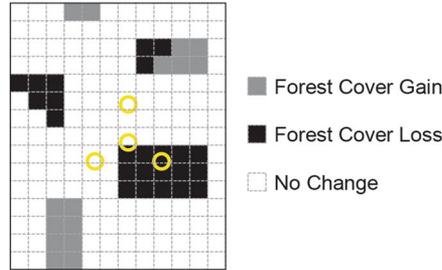
Land Use Change
Forest inventory measurement of land use at plot center two points in time.

Forest Cover Change
Gain and loss in forest cover (Hansen et al. 2013) combined for estimate of net gain/loss for each inventory plot.

Forest Cover Gain

Forest Cover Loss

No Change

Figure 1. Schematic of study methods for estimating forest land-use change and forest cover change at each forest inventory plot and subsequent scaling up to the eastern U.S. via study hexagons. Land-use change is photo interpreted at each forest inventory plot location based on high-resolution imagery (~1 m) at two points in time. Net forest cover change based on Landsat analysis (Hansen and others 2013) is estimated for the same inventory plots. The percentage of land-use change, forest cover change, and forest carbon stock change were then determined for each study hexagon (see ''Methods'' section).

Red hexagon denotes a study hexagon (observations). Black dots denote forest inventory plots.

Comparison
Land use change determined by forest inventory sample compared to net forest cover change through Landsat interpretation across study hexagons.

tree was considered downed dead wood when the lean angle of its central bole is greater than 45 degrees from vertical. Within each subplot, a 2.07-

m microplot offset 3.66 m from subplot center was established where only live trees with a diameter at breast height between 2.5 and 12.7 cm were

Table 1. Land-Use Classification, Definitions, and Number of Plots in This Study that were Used to Estimate Forest Land Use, Forest Cover, and Forest C Attributes with the Study Hexagons, 2002–2006 to 2007–2012, Eastern U.S

Specific land use category	Definition	Number of study sample points
Forest land	A land-use category that includes areas at least 36.6 m wide and 0.4 ha in size with at least 10% cover (or equivalent stocking) by live trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated	73,497
Agricultural land	Land managed for crops, pasture, or other agricultural use. Includes rangelands defined as areas composed of grasses, forbs, or shrubs. Includes other wetlands	80,373
Developed	Land used primarily by humans for purposes other than forestry or agriculture. Also includes nonvegetated land uses such as beaches, rock, and ice	16,335

inventoried. For complete details regarding the FIA sample design, plot protocols, and data management, please refer to Bechtold and Patterson (2005) and USDA (2014a, b, c).

Forest Carbon Estimation

It should be strongly noted that Good Practice Guidance and Guidelines (IPCC 2003, 2006) relevant to the development of NGHGIs for submission to the UNFCCC were not strictly followed in this study. The goal of this study was to evaluate the development of forest sector C baselines using land-use versus cover information. A complete accounting in terms of a NGHGI would require C estimates associated with all nonforest land uses at time one. Unfortunately, no field-based C inventory exists for these land uses that succinctly aligns with the forest inventory. The U.S. uses lower tiers (that is, less sophisticated and/or country specific) of the Good Practice Guidelines to accommodate this accounting requirement for the U.S. NGHGI (US EPA 2016). In this study, it was felt that such accommodation of lower tier methods with the millions of field-based tree observations in the forest sector would increase the uncertainty associated with study results rather than reduce them. Hence, field data (USDA 2014b) in this study were taken entirely from the FIA database (USDA 2014a) using the forest inventory in 37 states of the eastern U.S. (Figure 1) for a total of 170,205 plots first established between 2002–2006 and remeasured 5 years later from 2007–2012. Although the FIA annual inventory system is established across all coterminous states, the eastern states have been completely remeasured allowing for an empirical assessment of forest stock C change. Western states have only been remeasured in a few instances (Woodall and others 2015a). The associated data

are available for download at the following site: http://fiatools.fs.fed.us (FIA Datamart, USDA 2014c).

This study used a stock difference approach (IPCC 2003, 2006) as a surrogate for C flux where the total stock of C by component (for example, aboveground live biomass) was estimated at two points in time (at the plot level) with the difference divided by the remeasurement period (in years) serving as an estimate of average annual flux (C Mg ha^{-1} y^{-1}). An ecosystem approach was used for C sink/source nomenclature where positive values indicate sequestration (that is, assimilation from the atmosphere) and negative values indicate an emission. Additionally, the term "flux" refers to any movement of C either between pools (that is, lateral) or to the atmosphere. It should be strongly noted that a total net terrestrial change in C stocks was not evaluated in this study as would be required under Good Practice Guidance and Guidelines (see Land-Use Change and Forestry; IPCC 2003, 2006). Such an alignment with international guidelines for national-scale submission of greenhouse gas budgets to the UNFCCC would require estimates of C flux across all land uses beyond just forest, which was not the objective of this study. As such, study results should not be considered a critique of current terrestrial C accounting mechanisms (for example, IPCC 2003, 2006) as the goal of this study was an evaluation of the use of forest cover versus forest land-use assessments in the context of conclusions regarding forest C dynamics relevant to general forest C monitoring efforts.

Aboveground standing dead and live tree C stocks were calculated in this study using the Component Ratio Method (CRM, Woodall and others 2011). Briefly, the CRM facilitates the calculation of tree component biomass (for example, tops and limbs) as a proportion of the total above-

ground biomass based on component proportions from Jenkins and others (2003). For standing dead trees, which may lack some or all of the components calculated using CRM (for example, loss of limbs), structural and decay reduction factors were applied by decay class and species (Domke and others 2011). Standing dead/live total biomass was converted to C mass assuming 50% C content of woody biomass. Belowground estimates of coarse root C were not included as new estimation approaches are still in development (Russell and others 2015). The estimation of the soil organic C, forest floor, and downed dead wood stocks from each pool were accomplished using plot-level models as implemented in the U.S. NGHGI (Smith and others 2013; US EPA 2016). These C stocks are based on models using variables such as live tree C density and stand age with coefficients by geographic region and/or forest type. For example, soil organic C stocks are based on models parametrized using national databases of soil survey results (Amichev and Galbraith 2004). Finally, the dead wood pool was a combination of the standing dead tree C collected from the 7.32-m fixed-radius subplots combined with downed dead wood pool estimates based on field data collected on a subset of FIA forest plots (Domke and others 2013).

Joint Forest Use and Cover Analysis

To facilitate analysis at spatial scales suited for this study's sample intensity, forest C, forest land use, and forest cover were summarized by a hexagonal grid (not affiliated with smaller-sized hexagons used for forest inventory plot spatial distribution) developed for this study based on the FIA plot sampling intensity (Figure 1). The use of discrete hexagons appropriately sized (1384 km^2) for the FIA plot network sample intensity (median of 58 sample points per hexagon, for total plot counts see Table 1) facilitated visual interpretation of spatial patterns and evaluation of conclusions regarding forest C attributes in the context of land uses. Forest cover change area (derived directly from Hansen and others' (2013) publicly available data) estimated for each FIA plot was summed within each study hexagon to estimate a net forest cover change for the entire hexagon. As each hexagon was considered an observation in this study, forest C stocks, forest land-use change, and net forest cover change were estimated for each hexagon that had at least 8 sample points (that is, FIA plots), which excluded hexagons with a majority of their area outside the study area. This minimum sample size roughly corresponds to requirements for the

creation of strata during FIA's poststratification exercises for the estimation of population attributes (a minimum of 4–12 plots depending on forested conditions; Bechtold and Patterson 2005). A population estimate of total forest ecosystem C stocks at time one and time two was computed in addition to stock and stock change for each C pool. Forest land-use change was calculated as the change in the percent land use between time one and time two by land-use category. The C stock change across forest pools was computed at the plot level then scaled up to the hexagon based on the number of plots per hexagon at time 1. The use of the relatively large study hexagons with a substantial number of plot observations relieved privacy issues (that is, laws guaranteeing the privacy of forest owners) while mitigating measurement errors over time due to changes in classification personnel.

To fully evaluate differences between forest land-use change and cover change, the distribution of observations between forest land-use change and cover change was determined for hexagons with at least 10% forest land use (to avoid spurious results in landscapes with sparse forest). Spearman's correlation coefficients were estimated for these study hexagons. The correlation matrix included forest land-use change and cover change in addition to forest C stock (time 2: all remeasured FIA plots 2007–2012) and stock change by individual C pools. The response variables used in the Spearman's rank correlation were evaluated for spatial autocorrelation using Moran's I and Geary's C (Cressie 1993). Test results indicated high spatial autocorrelation for the forest C stock and stock difference estimates among neighboring hexagon centroids that greatly diminished at the state scale down to nearly nonexistent at the scale of the study area (entire eastern U.S.). Based on these findings and the scale of the study area, spatial autocorrelation was deemed to not affect results at a detectable level.

As initial analysis indicated some substantial differences between forest land use and cover estimates, a parsimonious approach was adopted to explore these results more deeply. First, stock estimates from the aboveground live tree C and soil organic C pools were selected as case studies, as they are the two largest stocks and provide a contrast between estimates more closely aligned within situ observations (aboveground live tree C modeled from individual tree measurements; Woodall and others 2011) versus those completely simulated based on site variables (soil organic C models based on forest type and region; Smith and others 2013). Second, given the variability associ-

ated with land-use change and C stock change, the relationship between cover change and land-use change was examined in a 2×2 matrix of outcomes: quadrant 1 (forest use increase and cover decrease), quadrant 2 (forest use increase and cover increase), quadrant 3 (forest use loss and cover loss), and quadrant 4 (forest use loss and cover increase). Using this 2×2 matrix approach regarding outcomes of forest land use and cover comparisons, study observations with at least 50% forest land use and greater than median difference between forest land use and cover for each quadrant were extracted for additional analysis ($\sim$14% of total observations). This data subset enabled a more detailed examination of differences in land use versus cover assessments in the context of forest C monitoring. For this dataset, mean and associated standard errors of stock change for the aboveground live tree and soil organic C pools were estimated by quadrant. Finally, in addition to the C stock difference approach used for estimating changes in forest C stocks, means and associated standard errors of some components of live tree change (that is, individual tree remeasurement over time: gross growth, harvest, and mortality) were also calculated by quadrant using this data subset of predominantly forested landscapes. It is important to note that this analysis of live tree components of change did not include a land-use change component. Thus, the summation of live tree gross growth, mortality, and harvest will not result in a complete accounting of live tree biomass. This analysis should elucidate some of the individual tree dynamics that underlay the broader change in forest C stocks but not match in terms of a complete forest C budget.

To quantitatively assess the sensitivity of both forest land use and land cover to key forest change metrics, we employed generalized boosted regression models (GBMs; Makler-Pick and others 2011). A GBM analysis allows one to quantify the sensitivity of variables to input variables. In our case, we were interested in quantifying how sensitive forest land use and forest cover were to variables such as aboveground live tree C and soil organic C stocks, percentage of polygons that were agriculture and developed, land cover (in the case for determining the sensitivity of land use), and land use (in the case for determining the sensitivity of land cover). In this machine learning algorithm, regression trees are calculated where each tree is designed to predict the residuals from the preceding tree. Of particular interest in a GBM analysis is the relative influence of each input parameter on model output. Relative influence is based on minimizing a loss function after splitting an input parameter within a regression tree and then averaging across all trees generated in the GBM. The relative influence metric for a specific input variable ranges from 0 (no influence) to 100 (complete influence), and the cumulative sum of relative influence scores totals 100.

The GBM method as described in Friedman (2001) was implemented for forest land use and forest cover using the 'gbm' package in R (Ridgeway 2013). Assuming a high correlation between land use and land cover, land cover would seemingly rank as a variable with high relative influence on land use, and vice versa. Each GBM was run using a squared error (Gaussian) distribution with three-way variable interactions and five-fold cross-validation. One thousand regression trees were run in total with one half of the data used for training the GBM.

RESULTS

Lack of Relationship between Changes in Forest Land Use versus Cover

Overall, the area of forest land use increased over this study's time period (Figure 2a). In contrast, the change in forest cover (Figure 2b) suggested widespread loss of cover concentrated in areas of the southern U.S., as well as portions of the northern Lake States (that is, recent Ham Lake and Pagami Creek fires in northeastern Minnesota) and northern New England. The correlation coefficient between forest land-use change and cover change was extremely weak at 0.08 (p value = 0.0002). Both forest land-use change and cover change were weakly correlated with C stocks (time 2) and stock change (Table 2). All significant (p value <0.05) correlation coefficients between forest land use/cover and forest C stocks were weakly negative, although forest cover correlations were somewhat stronger than those with forest LU. Correlations were stronger between forest land use/cover and forest C stock change (Table 2). For the stock change estimates associated with modeled forest C pools (for example, soil organic C), the correlation coefficients ranged from 0.25 to 0.56 with forest land-use change. In contrast, these correlations ranged from 0.09 to 0.19 for forest cover change (significant correlations, p value <0.05). One curious result was that the aboveground live tree C stock change was negatively (-0.08) correlated with forest land-use change but positively correlated with forest cover change (0.08) (for related discussion see Woodall and others 2015b).
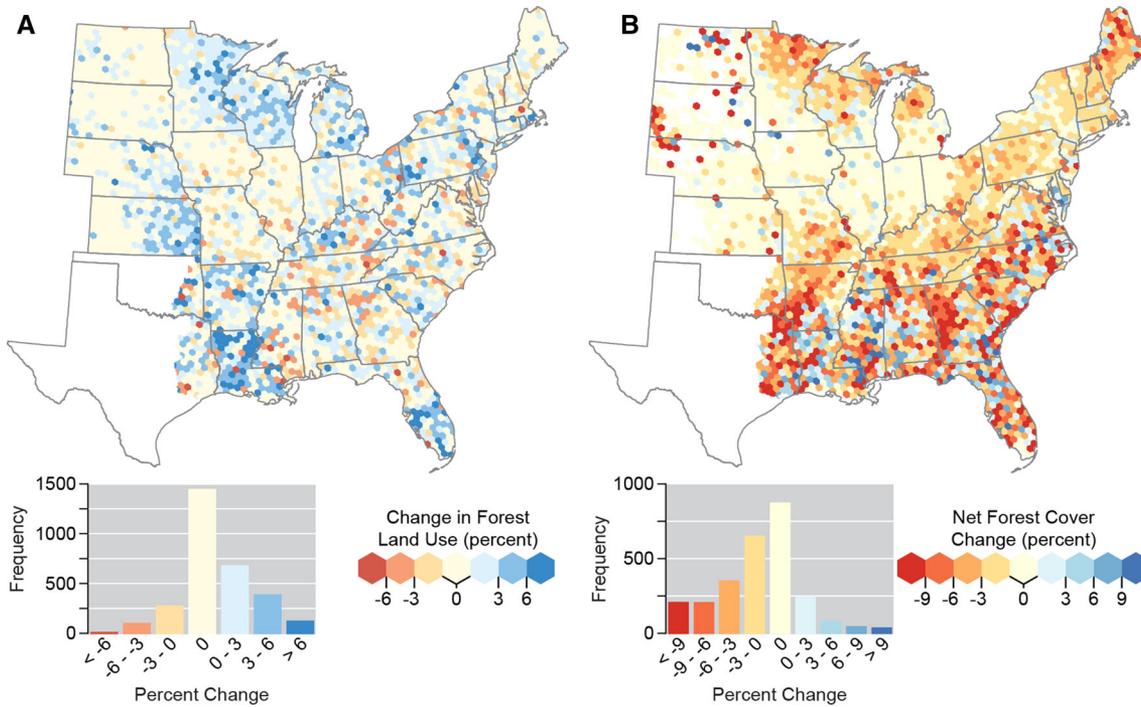
Figure 2. Percent change in **A** forest land use and **B** net forest cover change by study hexagons, 2002–2006 to 2007–2012, eastern U.S. Forest land use appears to increase in contrast to apparent widespread loss of forest cover.
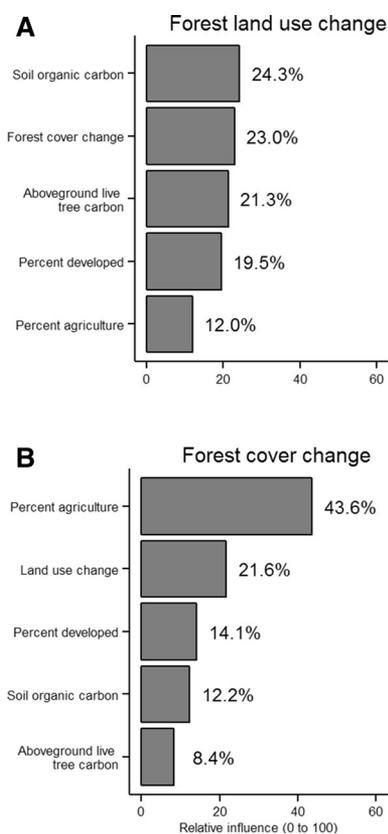
Table 2. Spearman Rank Correlation Coefficients Between Forest Carbon Stocks and Stock Change versus Land-use Change (time 2) and Land Cover Change, Eastern U.S. (for Study Hexagons Having at Least 10% Forest Land Use)

Forest carbon stocks (C Tg) and stock change by pool (C Tg y^{-1})	Forest land-use change (%)	Forest net land cover change (%)
Stocks		
Aboveground live tree	−0.13	−0.16
Downed dead wood	−0.10	−0.18
Litter	*−0.04*	−0.19
Soils	*0.01*	−0.19
Standing dead tree	−0.05	−0.18
Understory	−0.09	−0.24
Stock change		
Aboveground live tree	−0.08	0.08
Downed dead wood	0.32	0.13
Litter	0.28	0.10
Soils	0.54	0.09
Standing dead tree	0.25	0.19
Understory	0.56	*0.02*

Correlation coefficient between forest land use and cover change was 0.08 (p value = 0.0002)
Italicized coefficients indicate p value > 0.05

Sensitivity analyses (that is, GBMs) revealed that forest cover change was not the most influential variable describing forest land-use change (Figure 3). Instead, soil organic C stocks displayed the most influence on forest land-use change (relative influence of 24.3%), perhaps related to more productive soils at the interface of croplands and forests. Similarly for forest cover change, the

A

Forest land use change

Soil organic carbon	24.3%
Forest cover change	23.0%
Aboveground live tree carbon	21.3%
Percent developed	19.5%
Percent agriculture	12.0%

0   20   40   60

B

Forest cover change

Percent agriculture	43.6%
Land use change	21.6%
Percent developed	14.1%
Soil organic carbon	12.2%
Aboveground live tree carbon	8.4%

0   20   40   60
Relative influence (0 to 100)

Figure 3. Sensitivity results (boosted regression models) between **A** forest land-use change and **B** forest cover change versus a selection of study metrics, top five most sensitive metrics included, 2002–2006 to 2007–2012, eastern U.S.

percentage of agriculture within a study polygon was the most influential variable describing forest cover change (relative influence of 43.6%), while forest land-use change displayed the second-highest influence on cover change (21.6%). Perhaps most importantly, cover change is not the highest ranking variable influencing land-use change and vice versa. It should be noted that these GBM results do not indicate the strength of a variable for directly estimating C stock change. Instead, they estimate the influence of a variable relative to all other considered variables in terms of estimating forest C stock dynamics.

Forest Carbon Stocks Increase Despite Losses in Forest Cover

Given the apparent difference in correlations between forest land use/cover and field-based biomass C stock change versus modeled forest soil C stock change, aboveground live tree C and soil organic C stock change were examined in more

detail. Across the eastern U.S. aboveground live tree C stocks increased during the study period with a strong right skew towards sequestration (Figure 4a). In contrast, stock change in the soil organic C pool was more normally distributed about zero with a slight right skew towards sequestration (Figure 4b).

The distribution (along a 1:1 line that indicates perfect alignment) in forest land use versus cover by the C sink status of stock increase (Figure 5a; $n = 545$ hexagons) or stock decrease (Figure 5b; $n = 1732$ hexagons) for a combination of the aboveground live tree and soil organic C pools was examined by scatterplots. In a manner similar to the correlation results, there was no apparent relationship between changes in forest land use versus forest cover. There was an obvious skewing (76% of hexagons) towards the more common observation of forest C stock increase, forest cover decrease, and forest land-use increase (upper left quadrant, Figure 5b). To further examine cases where land-use and cover changes were most divergent in the context of forest C stock change, aboveground live tree and soil organic C were examined within a 2×2 matrix of land-use change and cover change scenarios for observations with at least 50% forest land use and a greater than median difference in use and cover (Figure 6). Using this dataset, the spatial distribution of quadrants of land use versus cover scenarios appeared random. The majority of hexagons had a scenario of net forest cover decrease and forest land use increase ($n = 235$) in contrast to the second most common observation, which was net forest cover decrease and land-use decrease ($n = 102$).

Carbon Dynamics Conclusions within the Context of Forest Land Use versus Cover Metrics

Means of aboveground live tree and soil organic C stock change (Figure 7) by quadrant (Figure 6) indicated a series of issues can occur if one attempts to infer forest C dynamics across large scales. For example, if cover was solely used to detect land-use change, regional patterns associated with aboveground live tree and soil organic C would suggest that a decrease in forest cover led to an increase in these stocks (Figure 7a). Second, a loss in both cover and use indeed results in a reduction in aboveground live tree C stocks but oddly resulted in a potentially spurious increase in soil organic C stocks due to a state change (that is, change in one of the stock model's parameters or censored
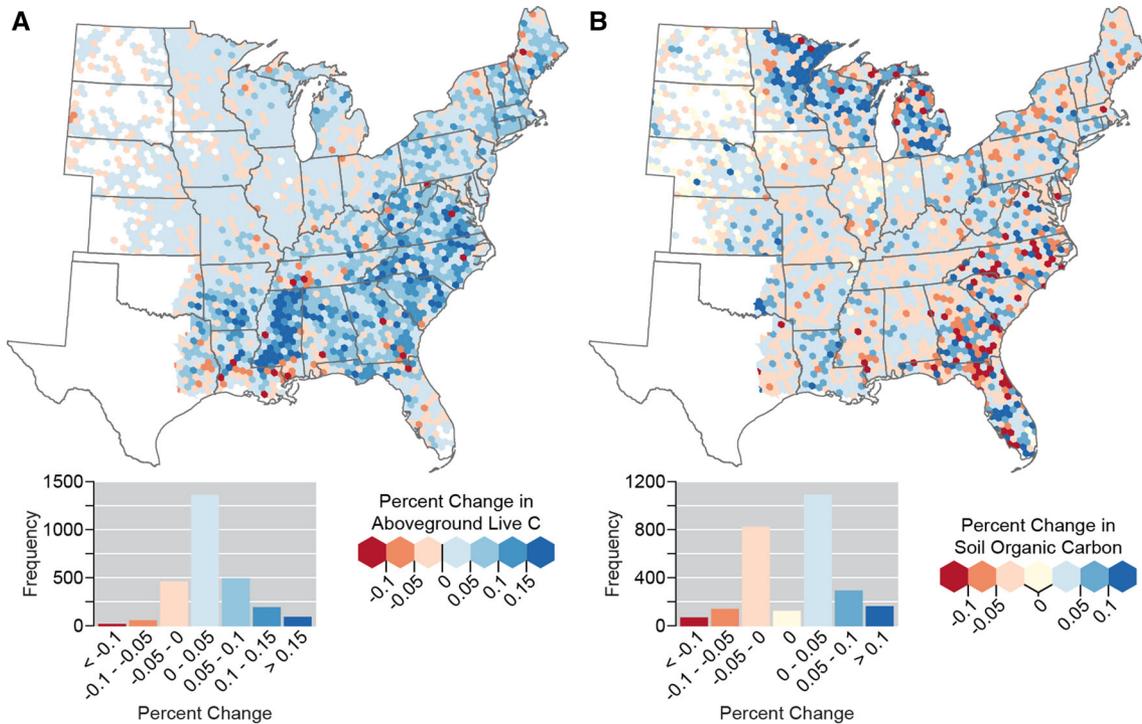
Figure 4. Percent change in **A** aboveground live and **B** soil organic carbon stocks in forests, 2002–2006 to 2007–2012, eastern U.S. Aboveground live tree carbon stocks appear to increase over much of the U.S., whereas soil organic carbon stocks appear steady-state except for increases across the Lake States.

observation of nonforest land-use soil organic C stocks). Third, the same effect was seen in the results of quadrant four (forest use loss but cover increase) but with larger C stock changes.

To evaluate forest C dynamics more deeply, live tree volume components of change were evaluated by quadrant, which is an alternative estimation approach to stock change (Figure 7b). The highest mean harvest removals of live tree volume were found in the land-use loss and cover loss scenario (that is, quadrant three). In addition, for this scenario, there was a net loss of forest live tree volume with harvest removals and mortality exceeding gross growth. In contrast, the highest increase in volume growth was for the scenario in which both forest land use and cover increase (quadrant two). The most noteworthy scenario is forest use increase and cover decrease (quadrant one), which was the most common scenario across the eastern U.S. Under this scenario, a cover decrease did not reduce aboveground live tree C stocks, rather these stocks increased while land use increased concomitantly. In addition, this scenario demonstrated positive net live tree volume growth even when accounting for deductions in mortality and harvest removals (where forest land use can remain).

DISCUSSION

Land cover assessments, while often the only viable alternative when empirical land-use change information is absent, did not directly correlate with observed land-use change in this study. A similar finding was reported by Coulston and others (2014) who compared forest cover versus forest land use and speculated that such a divergence may have serious implications regarding forest C monitoring and policy development. In terms of C, our study found stock changes in the pools of total forest ecosystem C (all pools), downed dead wood C, and soil organic C were much more strongly correlated with forest land-use change than net forest cover change. When comparing forest land-use change to forest cover change, the most common situation was for net forest cover change to be negative in contrast to the forest land-use assessment, indicating increased forest land use. Holmgren (2015) recently reported a similar finding using forest land-use assessments from The Food and Agriculture Organization of the United Nations' Global Forest Resources Assessment (MacDicken and others 2015) and forest cover monitoring from the Global Forest Watch (GFW 2015).
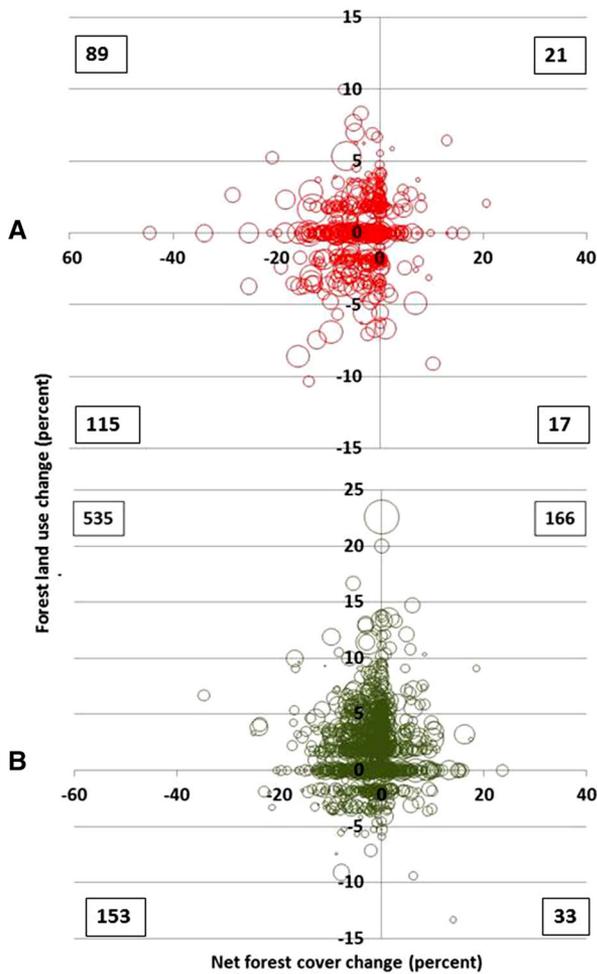
Figure 5. Changes in proportions of forest land use versus forest land cover by study hexagon for **A** net forest carbon decrease and **B** net forest carbon increase, 2002–2006 to 2007–2012, eastern U.S. Nonzero observation counts provided for each quadrant in a box (land use versus cover factorial design). Hexagon must include at least 10% forest land use. Observations sized by carbon stock change (forest carbon pool = aboveground live tree + soil organic carbon).

Certainly in the context of Good Practice Guidance and Guidelines (IPCC 2003, 2006), there is not an a priori expectation of a net C reduction when a net forest cover loss occurs over longer time steps due to the growth rates on remaining undisturbed stands in the landscape coupled with varying levels of disturbance intensity (that is, minor forest stand perturbations) and net cover loss (that is, defoliation events). However, actively managed forests will experience substantial reductions in tree cover through cycles of stand treatments (Oliver and Larson 1996) for timber production (for example, Fox and others 2007) and other management objectives such as restoration (Larson and others
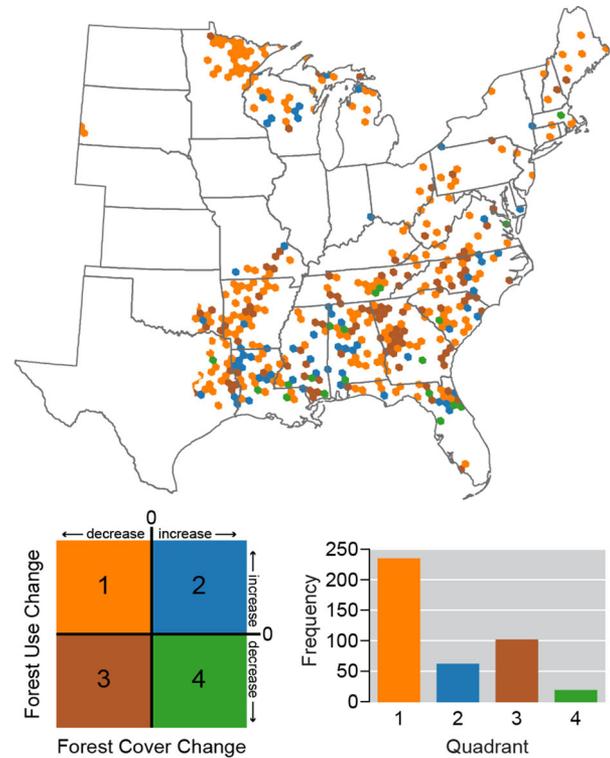

Figure 6. Distribution of forest land use versus land cover hexagons where the forest land use ≥50% and the difference between land use and cover exceeds the median of all observations for the quadrant of observations between land use and change, 2002–2006 to 2007–2012, eastern U.S. The most common observation was that of an increase in forest land use and a decrease in forest cover.

2012) and similar transient reductions will occur in natural forests impacted by natural disturbance events (Kurz and others 2008). Therefore, a divergence between a forest cover assessment (Hansen and others 2013) and FIA's land-use assessment (for example, Coulston and others 2014; Holmgren 2015) was identified in landscapes with rates of net volume growth even when accounting for mortality and harvest (that is, higher rates of C sequestration). We hypothesize that such a divergence creates a bias in land-use assessments (for example, C monitoring) that rely solely on land cover metrics in landscapes with managed forests. Such a bias can be described as an overattribution of forest C change to land-use change rather than correct attribution to forest management activities (that is, forests remaining forests). Further, it may create conflicting and spurious trends, such as observing high rates of C sequestration in landscapes with relatively high levels of net cover loss over short time steps due to aforementioned forest management activities.
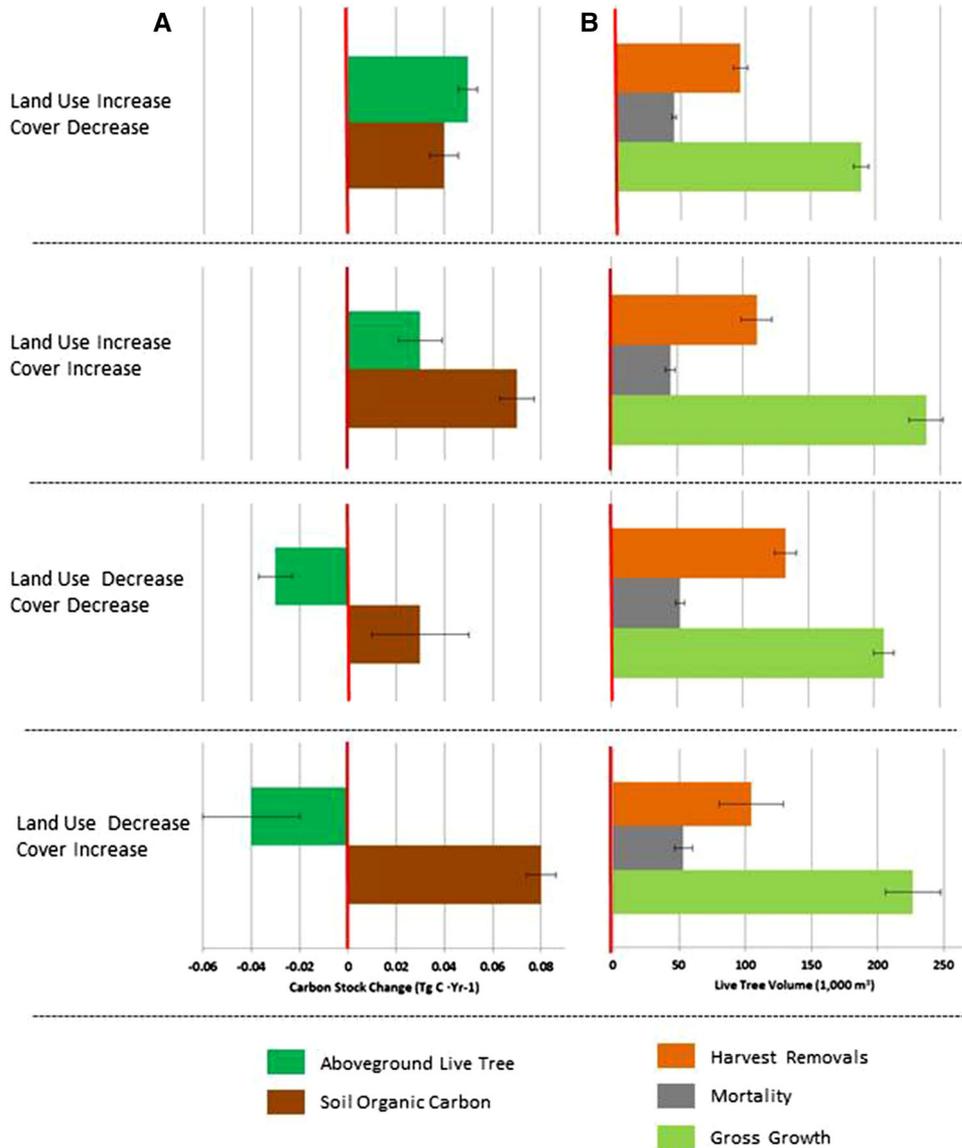
Figure 7. Mean and associated standard errors of **A** aboveground live tree and soil organic carbon stock change (positive value = sequestration; Tg C y^{-1}) and **B** components of live tree volume change (thousand m^3) for matrix of land use versus land cover analysis (*quadrant 1* forest use increase, cover decrease; *quadrant 2* forest use increase, cover increase; *quadrant 3* forest use loss, cover loss; *quadrant 4* forest use loss, cover increase), 2002–2006 to 2007–2012, eastern U.S. (*Note* individual tree volume components of change do not include land-use change and subsequently do not represent a complete accounting).

Forest land-use metrics should not be considered superior to cover assessments as they are inherently different (for example, intended use of land versus vegetative cover) with their own sources of error and varying spatial resolutions. In the case of this study's land-use assessment, there is error in high-resolution aerial photo interpretation and landowner intent assessment. In the case of Landsat imagery, there is cover classification error and coarse spatial resolution (30 m). Additionally, although care was given to align the Landsat acquisition dates of the Hansen and others (2013) product with the forest inventory plot measurements, we expect unavoidable discrepancies at the pixel scale could increase differences between land use and land cover. Despite this, the results of this study still suggest that the two cannot be used interchangeably as if they are synonymous. As examined in the sensitivity analysis, using forest cover change as a surrogate for forest land-use change (and vice versa) does not result in the highest relative influence on land-use metrics (for example, percent agriculture and developed) and C stocks (for example, aboveground live tree and soil organic C). The impact of adopting either approach in the context of conclusions regarding forest C flux should be considered by those monitoring forest C. A number of stand-level processes can be hypothesized as potential drivers in the divergences between forest cover versus use assessments (Figure 8). Outcomes such as forest use/cover increasing are intuitive such as old field succession. Other outcomes are not nearly as intuitive such as a decrease in forest use but an increase in cover. In cases such as this, there may be less

Forest Use Decrease / Forest Cover Increase — Quadrant 1

Drivers: Activities that convert forest to non-forest land use but may have a delay in tree mortality or retain tree canopy cover below minimum for classification as a forest (e.g., flooding of forest sufficient to kill most trees)

Flooded ash forest, Minnesota

Forest Use Increase / Forest Cover Increase — Quadrant 2

Drivers: Either active conversion or abandonment of non-forest land uses to forest (e.g., old field succession)

Old field succession, Indiana

Forest Use Decrease / Forest Cover Decrease — Quadrant 3

Drivers: Either active conversion or disturbance induced conversion of forest to non-forest (e.g., volcanic explosion or conversion to settlements)

Clear cut and conversion to grassland, Florida

Forest Use Increase / Forest Cover Decrease — Quadrant 4

Drivers: The conversion of non-forest land use to forest and/or regenerating forests with insufficient forest canopy cover to be detected by satellite while remaining a forest land use

New pine plantation, Florida

Figure 8. Examples of potential stand-level drivers of various combinations of forest cover and land-use change results. Divergences between changes in forest use versus forest cover assessments appear to stem from the inability of coarse resolution (30 m) remotely sensed imagery to detect forest regeneration practices and/or landowner intent, the extended latency of land-use surveys to detect disturbance events, and the measurement/detection error associated with both land use and cover monitoring efforts.

influence of stand-level drivers and more impact of measurement/detection errors combined with latency (for example, Figure 8, recent submergence of a forest or recent afforestation of cropland). Neither metric needs to be considered in isolation as both can aid with forest monitoring and management. Better forest ecosystem C monitoring and management will not only need to incorporate remotely sensed products such as those derived from NAIP interpretation (such as the land-use assessment in this study) and Landsat cover detection (that is, Hansen and others 2013) but also in situ observations of forest C stocks as they transition in and out of the forest land use. These observations will be especially important for monitoring dynamic forest landscapes where widespread active management is applied (for example, harvesting), such as in the southeastern U.S. (Oswalt and others 2014), or where large-scale, high-severity natural disturbance events (for example, insects or fire) are more frequent, such as in boreal systems (Kurz and others 2008).

An additional complexity to be resolved is that of more empirically derived forest C estimates (that is, allometric tree biomass models dependent on species, diameter, and height) versus simulated C estimates in the context of land-use change versus cover change assessment (not the case in process-based modeling exercises). Modeled C stock change estimates (for example, soil organic C, Guo and Gifford 2002) may be considered spurious in the context of some land-use change situations as they are sensitive to state change (for example, levels of aboveground live tree biomass or forest type; Smith and others 2013). In our study, we found the strongest correlations between the land use/cover metrics and forest C pools more dependent on simulation exercises (for example, soil organic C). In terms of soil organic C, certainly a transition from forest to settlement land uses does not result in an immediate release of a substantial proportion of soil C due to harvest and land conversion (Nave and others 2010). Such an idiosyncrasy is a weakness of empirical observations of land-use change if

forest C pool models do not explicitly incorporate land-use change metrics. In contrast, many of the regions identified by cover assessments as experiencing forest land-use reductions have strategic forest management regimes that have focused on increasing levels of aboveground biomass over the past several decades (Fox and others 2007), highlighting the difficulty in equating cover assessments with broad C dynamics. Furthermore, cover assessments may not detect small-scale mortality events (for example, drought-induced mortality) or selective logging practices in which individual trees contributing disproportionately to the aboveground C stocks are removed or die, yet forested conditions remain largely intact (Asner and others 2005). Regardless of whether empirical observations or modeled land use from cover assessments are used in C monitoring efforts, explicit alignment with forest C monitoring techniques is needed to reduce the associated uncertainty.

Future research should explore how to better integrate both land cover and land-use assessments not only to benefit forest C monitoring and reporting (for example, NGHGIs) but also to more fully inform the management of forest C across landscapes (for example, policies regarding afforestation and land-use planning). Which remote sensing products might be created that incorporate empirical land-use observations? Perhaps the use of much finer resolution space-based imagery (if a 1990 base line year is not needed) coupled with refined classification algorithms would one day replicate the land-use assessment used in this study (that is, human interpretation of high-resolution aerial imagery verified by a forest inventory). Mascorro and others (2015) indeed suggest that increased spatial resolution of remotely sensed imagery should be the first priority when monitoring forest C. Full integration of forest C stock and stock change estimates based on in situ measurements (Domke and others 2013; Domke and others unpublished) may reduce or eliminate spurious results (Woodall 2012) such as those observed in the soil organic C pool in this study. Taken together, perhaps the greatest reductions in forest C monitoring uncertainty can be gained through refining the C stock estimation of the largest stocks (for example, soil organic C, Woodall and others 2013) in concert with improved cover assessments using empirical land-use observations (Brown and others 2013) and finer spatial resolution imagery (Mascorro and others 2015). The central theme of such an endeavor would be the increased collection and use of in situ observations coupled with increased transparency to enable wider adoption and review.

CONCLUSIONS

When constructing a forest C inventory there are a myriad of monitoring approaches to be considered that can profoundly affect C resource conclusions and associated policies. Such monitoring decisions can involve whether land use or land cover is used to monitor landowner intent (for example, high-resolution aerial imagery interpretation versus vegetative cover mapping), which modeling procedures are used to estimate C pools (for example, tree-level biomass models versus site-level soil organic C models), and which C accounting approach is adopted (for example, net versus gross land-use area change). Our study found very little correlation between forest land-use change and net forest cover change, while the strongest correlations with forest C were identified between forest C stocks based on coarse models (for example, soil organic C) and forest land-use change. When comparing forest land use to cover in the context of forest C monitoring, the most common observation was that of forest C stock increase with the forest cover assessment indicating a cover loss while forest land use increased. For some landscapes (that is, 1384 km^2 hexagons) in our study, if cover was used solely to detect land-use change one would conclude that a decrease in forest cover led to an increase in aboveground live tree and soil organic C stocks. Furthermore, in such a situation, we found positive net live tree volume growth even when accounting for deductions in mortality and harvest removals. In light of these stark differences between forest land-use change and cover change metrics, we suggest that they should not be used interchangeably as if synonymous. In addition, neither metric is superior as they are sensitive to different forest and land-use dynamics over varying timescales (for example, active forest management versus deforestation), indicating their application is dependent on information needs (for example, forest sustainability assessment versus national C accounting). Perhaps neither land-use nor cover change information in isolation provides sufficient activity data to estimate landscape-scale forest C stock change. In this tale of two different forest C inventories, the epilogue is that these C inventories are for the same forest but with different authors (that is, land use versus cover), which can result in starkly contrasting conclusions in regard to forest C dynamics and true land use.

REFERENCES

Amichev BY, Galbraith JM. 2004. A revised methodology for estimation of forest soil carbon from spatial soils and forest inventory data sets. Environ Manag 33(Suppl. 1):S74–86.

Asner GP, Knapp DE, Broadbent EN, Oliveira PJC, Keller M, Silva JN. 2005. Selective logging in the Brazilian Amazon. Science 310:480–2.

Bechtold WA, Patterson PL. (Eds.), 2005. The enhanced forest inventory and analysis program—national sampling design and estimation procedures. USDA Forest Service General Technical Report SRS-80. Asheville, NC.

Birdsey R, Pregitzer K, Lucier A. 2006. Forest carbon management in the United States: 1600–2100. J Environ Qual 35:1461–9.

Brown DG, Robinson DT, French NHF. 2013. Perspectives on land-change science and carbon management. In: Brown, DG and others, Eds. Chapter 22 in land use and the carbon cycle. Cambridge University Press, Cambridge.

Caspersen JP, Pacala SW, Jenkins JC, Hurtt GC, Moorcroft PR, Birdsey RA. 2000. Contributions of land-use history to carbon accumulation in U.S. forests. Science 290:1148–51.

Cochran WG. 1977. Sampling techniques. New York: Wiley.

Coulston JW, Wear DN, Vose JM. 2015. Complex forest dynamics indicate potential for slowing carbon accumulation. Sci Rep 5:8002.

Coulston JW, Reams GA, Wear DN, Brewer CK. 2014. An analysis of forest land use, forest land cover and change at policy-relevant scales. Forestry 87:267–76.

Cressie NA. 1993. Statistics for spatial data, revised edition. Wiley. 928 p.

Domke GM, Woodall CW, Smith JE. 2011. Accounting for density reduction and structural loss in standing dead trees: implications for forest biomass and carbon stock estimates in the United States. Carbon Balance Manag 6:14.

Domke GM, Woodall CW, Walters BF, Smith JE. 2013. From models to measurements: comparing down dead wood carbon stock estimates in the U.S. forest inventory. PLoS ONE 8:e59949.

Domke GM, Walters BF, Perry CH, Woodall CW, Russell MB, Smith JE. In: Review. A framework for estimating litter carbon stocks in forests of the United States. Science of the Total Environment.

Ellis EC, Goldewijk KK, Siebert S, Lightman D, Ramankutty N. 2010. Anthropogenic transformation of the biomes, 1700 to 2000. Glob Ecol Biogeogr 19:589–606.

Environmental Protection Agency (US EPA) 2016. Forest sections of the land use, land-use change, and forestry chapter, and annex. In: US Environmental Protection Agency, Inventory of US Greenhouse Gas Emissions and Sinks: 1990–2014.

EPA 430-R-16-002. https://www3.epa.gov/climatechange/ghgemissions/usinventoryreport.html. Accessed 19 April 2016.

EOP. 2013. Executive office of the president: the president's climate action plan. Climate Action Plan. http://www.whitehouse.gov/sites/default/files/image/president27sclimateactionplan.pdf. Accessed March, 2015.

Foster DR. 1992. Land-use history (1730-1990) and vegetation dynamics in central New England, USA. J Ecol 80:753–71.

Fox TR, Jokela EJ, Allen HL. 2007. The development of pine plantation silviculture in the southern United States. J Forest 105:337–47.

Friedman JH. 2001. Greedy function approximation: a gradient boosting machine. Ann Stat 29:1189–232.

Gibbs HK, Brown S, Niles JO, Foley JA. 2007. Monitoring and estimating tropical forest carbon stocks: making REDD a reality. Env Res Lett 2:045023.

GFW. 2015. Global Forest Watch. http://www.globalforestwatch.org/. Accessed November 23, 2015.

Guo LB, Gifford RM. 2002. Soil carbon stocks and land-use change: a meta-analysis. Glob Change Biol 8:345–60.

Hansen MC, Potapov PV, Moore P, Hancher SA, Turubanova A, Tyukavina A, Thau D, Stehman SV, Goetz SJ, Loveland A, Kommareddy A, Egorov A, Chini L, Justice CO, Townshend JRG. 2013. High-resolution global maps of 21st—century forest cover change. Science 342:850–3.

Holmgren P. 2015. Can we trust country-level data from global forest assessments? For Source 20:8–9.

Houghton RA, Hackler JL, Lawrence KT. 1999. The U.S. carbon budget: contributions from land-use change. Science 5427:574–8.

Houghton RA. 2003. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and management 1850–2000. Tellus 55B:378–90.

IPCC. 2003. In: Penman J, Gytarsky M, Krug T, Kruger D, Pipatti R, Buendia L, Miwa K, Ngara T, Tanabe K, Wagner F, Eds. Intergovernmental panel on climate change: good practice guidance for land use, land-use change, and forestry. IGES, Japan.

IPCC. 2006. Intergovernmental panel on climate change: guidelines for national greenhouse gas inventories: Volume 4 Agriculture, Forestry and Other Land Use. In: Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K, Eds. Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan.

Jenkins JC, Chojnacky DC, Heath LS, Birdsey RA. 2003. National scale biomass estimators for United States tree species. Forest Science 49:12–35.

Kurz WA, Dymond CC, White TM, Stinson C, Shaw CH, Rampley GJ, Smyth C, Simpson BN, Neilson ET, Trofymow JA, Metsaranta J, Apps MJ. 2009. CBM-CFS3: a model of carbon-dynamics in forestry and land-use change implementing IPCC standards. Ecol Model 220:480–504.

Kurz WA, Stinson G, Rampley GJ, Dymond CC, Neilson ET. 2008. Risk of natural disturbances makes future contribution of Canada's forests to the global carbon cycle highly uncertain. PNAS 105:1551–5.

Larson AJ, Stover KC, Keyes CR. 2012. Effects of restoration thinning on spatial heterogeneity in mixed-conifer forest. Can J For Res 42:1505–17.

MacDicken K, Jonsson Ő, Piňa L, Maulo S, Adikari Y, Garzuglia M, Lindquist E, Reams G, D'Annunzio R. 2015. The global forest resources assessment 2015: how are the world's forests

changing?. Rome: Food and Agriculture Organization of the United Nations.

Makler-Pick V, Gal G, Gorfine M, Hipsey MR, Carmel Y. 2011. Sensitivity analysis for complex ecological models—a new approach. Environ Modell Softw 26:124–34.

Mascorro VS, Coops NC, Kurz WA, Olguín M. 2015. Choice of satellite imagery and attribution of changes to disturbance type strongly affects forest carbon balance estimates. Carbon Balance Manag 10:30.

Nabuurs G-J, Lindner M, Verkerk PJ, Gunia K, Deda P, Michalak R, Grassi G. 2013. First signs of carbon sink saturation in European forest biomass. Nat Clim Change 3:792–6.

NAIP. 2015. National Agriculture Imagery Program. U.S. Department of Agriculture, Washington, DC. http://www.fsa. usda.gov/programs-and-services/aerial-photography/imagery-programs/naip-imagery/. Accessed November 23, 2015.

Nave LE, Vance ED, Swanston CW, Curtis PS. 2010. Harvest impacts on soil carbon storage in temperate forests. For Ecol Manag 259:857–66.

Nowacki GJ, Abrams MD. 2015. Is climate an important driver of post-European vegetation change in the eastern United States? Glob Change Biol 21:314–34.

Oliver CD, Larson BC. 1996. Forest stand dynamics. New York: McGraw Hill.

Oswalt SN, Smith WB, Miles PD, Pugh SA. 2014. Forest resources of the United States, 2012: a technical document supporting the Forest Service 2015 update of the RPA Assessment. General Technical Report WO-91. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office.

Pan Y, Birdsey RA, Fang J, Houghton R, Kauppi PE, Kurz WA et al. 2011. A large and persistent carbon sink in the world's forests. Science 333:988–93.

Radeloff VC et al. 2012. Economic-based projections of future land use in the conterminous United States under alternative policy scenarios. Ecol Appl 22:1036–49.

Rhemtulla JM, Mladenoff DJ, Clayton MK. 2009. Historical forest baselines reveal potential for continued carbon sequestration. PNAS 106:6082–7.

Ridgeway G. 2013. GBM: generalized boosted regression models. R package version 2.1. Available from: http://CRAN.R-project.org/package=gbm.

Russell WB, Domke GM, Woodall CW, D'Amato AW. 2015. Comparisons of allometric and climate-derived estimates of tree coarse root carbon in forests of the United States. Carbon Balance Manag 10:20.

Smith JE, Heath LS, Hoover CM. 2013. Carbon factors and models for forest carbon estimates for the 2005–2011 National Greenhouse Gas Inventories of the United States. For Ecol Manag 307:7–19.

United Nations Framework Convention on Climate Change. 2013. Report on the individual review of the inventory submission of the United States of America submitted in 2012. FCCC/ARR/2012/USA.

USDA 2011. National report on sustainable forests, 2010. Washington, DC: U.S. Department of Agriculture Forest Service, Washington. FS-979.

USDA 2014a. The forest inventory and analysis database: database description and user guide for phase 2 (version 6.0.1). http://www.fia.fs.fed.us/library/database-documentation/current/ver6.0/FIADB%20User%20Guide%20P2_6-0-1_final.pdf (2014).

USDA. 2014b. Forest inventory and analysis national core field guide, volume i: field, data collection procedures for phase 2 plots, version 6.1. Washington, DC: U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis, Washington, D.C.: URL: http://www.fia.fs.fed.us/library/. Accessed February, 2015.

USDA. 2014c. Forest inventory and analysis national program—data and tools—FIA data mart, FIADB Version 5.1. Washington, DC: U.S. Department of Agriculture, Forest Service. http://apps.fs.fed.us/fiadb-downloads/datamart.html. Accessed June 24, 2014.

Westfall JA, Patterson PL, Coulston JW. 2011. Post-stratified estimation: within-strata and total sample size. Can J For Res 41:1130–9.

White D, Kimerling AJ, Overton WS. 1992. Cartographic and geometric components of a global sampling design for environmental monitoring. Cartogr Geogr Inf Syst 19:5–22.

Woodall CW, Heath LS, Domke GM, Nichols MC. 2011. Methods and equations for estimating volume, biomass, and carbon for trees in the U.S. forest inventory, 2010. US Forest Service General Technical Report NRS-GTR-88.

Woodall CW. 2012. Where did the U.S. forest biomass/carbon go? J For 110:113–14.

Woodall CW, Domke GM, Riley K, Oswalt CM, Crocker SJ, Yohe GW. 2013. Developing a framework for assessing global change risks to forest carbon stocks. PLoS ONE 8:e73222.

Woodall CW, Coulston JW, Domke GM, Walters BF, Wear DN, Smith JE, Anderson H-E, Clough BJ, Cohen WB, Griffith DM, Hagan SC, Hanou IS, Nichols MC, Perry CH, Russell MB, Westfall JA, Wilson BT. 2015a. The US Forest Carbon Accounting Framework: Stocks and Stock Change, 1990–2016. US Forest Service General Technical Report NRS-GTR-154.

Woodall CW, Walters BF, Coulston JW, D'Amato AW, Domke GM, Russell MB, Sowers PA. 2015b. Monitoring network confirms land use change is a substantial component of the forest carbon sink in the eastern United States. Sci Rep 5:17028.