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ORIGINAL PAPER

Effects of Forest Management and Land Use Change on Soil Carbon in the US Gulf Coastal Plain

Lucas E. Nave[1] · Kendall DeLyser[2] · Grant M. Domke[3] · Scott M. Holub[4] · Adrienne B. Keller[1] · Darren Miller[5] · Kevin A. Solarik[6] · Erik Schilling[5] · Jason Vogel[7] · Christopher W. Swanston[8]

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Abstract

The US Gulf Coastal Plain is a region of diverse forests and management systems and dynamic land use changes. These drivers affect carbon (C) stocks at ecosystem to regional scales, in turn affecting processes such as forest regeneration, hydrologic and biogeochemical cycling, and social and economic systems. We quantified land use change and forest management effects on C stocks in the Gulf Coastal Plain, focusing on soils, using two complementary approaches: meta-analysis of published literature and analysis of forest inventory data. Land use change had larger effects than forest management on soil C stocks at all scales. The direction and magnitude of change in soil C stocks differed among various types of land use change. From a regional perspective, the most important significant changes were those occurring with transfers between agricultural and forested land uses, though less extensive land uses also registered significant changes (e.g., wetland restoration, mine reclamation). Management of continuously forested land had subtler effects, which were focused on aboveground biomass and soil organic horizons. Organic horizons were consistently diminished by prescribed fire and may be reduced by harvesting, though evidence is limited for the latter, and declines are temporary in both cases. Aboveground activities intended to increase stand re-establishment and tree growth, such as competing vegetation control and fertilization, are at least compatible, and potentially synergistic with C storage in belowground pools. Overall, our results demonstrate how forest management can support forest C management and offer quantitative insights into specific management activities to aid researchers, policymakers, and practitioners.

Keywords Forest management · Soil carbon · Meta-analysis · National forest inventory · Southern pine

Extended author information available on the last page of the article

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Introduction

The US Gulf Coastal Plain is a disproportionately important region to the forest sector carbon (C) budget at national and even global scales. The highly productive forests of the region currently comprise over half of the total harvested wood output of the US as a whole, with wood products from southern pine (*Pinus* spp.) forest types dominating (Oswalt et al. 2019). While most of this fiber is used for lumber, pulp, and manufactured wood products, a substantial and growing share of it is used for energy production, from local mills across the region to global markets (Abt et al. 2014). In this region, the forest products sector supports hundreds of thousands of jobs and generates over 100 billion dollars annually in direct economic output (Dahal et al. 2015). Yet these impressive statistics speak only to what is removed from these forests. Within these ecosystems, per-acre forest growth rates are the most rapid, and standing stocks of live tree biomass carbon (C) are larger in the US south than in any other reporting region monitored by the USDA Forest Service, Forest Inventory and Analysis program (USDA Forest Service, FIA; Domke et al. 2021). Furthermore, as in most regions and forest types of the US, soils hold the majority of the total ecosystem C. In the Gulf States and the broader southern region, mineral soils hold 49% of the forest ecosystem C, aboveground biomass 37%, belowground biomass 7%, dead wood 4%, and forest floors 4% (Walters et al. 2024).

The large standing stocks and rapid C uptake in forests of the Gulf Coastal Plain mean that even modest per-acre effects of land use change and management in the region could have outsized impacts on the C budget of the region. Numerous assessments have examined how land use and management interact with drivers, such as disturbances and product end-uses, to affect the regional forest sector C budget, with differing system boundaries and assumptions frequently influencing their overall results (Gu et al. 2019; Henderson et al. 2024; Wear and Coulston 2015). Typically, such large-scale assessments explicitly address only the aspects of the ecosystem visible aboveground: tree biomass, land cover, and their rates of change. However, given the dominant size of the soil C pool in the forests of the Gulf States, and the likelihood that this C pool is not static, there is a need to account for soil C changes with land use change and management for a more comprehensive perspective on the forest sector C budget of the region (Cohen et al. 2008; Nave et al. 2022a; Sharma et al. 2024; Xiong et al. 2014).

We used synthesis methods applied previously in six other ecoregions of the US (Nave et al. 2019; 2021; 2022a; 2022b; 2024; 2025), drawing on multiple independent data sources to address the question: how do land use change and forest management affect soil C stocks in the Gulf Coastal Plain? Our work across other regions has shown that generalizations from large-scale meta-analyses often do not extend to the unique physiographic conditions, forest and soil types, and management systems of distinct regions, while site-level studies often struggle to scale their inferences to the broader landscape and regional extents where management is planned and practiced. This analysis remains focused on soil C stocks as the variable of principal interest, while also drawing in data on aboveground biomass due to its major importance to the forest C balance, and on soil bulk density as a proxy for soil physical disturbance.

Methods

Study Area

We delineated our study area according to the USDA Forest Service (Forest Service) ECOMAP system (Cleland et al. 1997; McNab et al. 2007). Our focus was on the Gulf Coastal Plain and lower Mississippi Valley, as constrained by the boundaries of our two previously published assessments focused on the South Atlantic States and the Central Hardwoods (Nave et al. 2022a; 2025). As such, the data that we synthesized were from 15 ECOMAP Ecological Sections encompassing portions of the states of Georgia, Florida, Alabama, Mississippi, Louisiana, Arkansas, Texas, Oklahoma, Missouri, Illinois, and Tennessee (Fig. 1). Ecological Sections tier beneath the Province level in the hierarchical ECOMAP system and are summarized below.

The climate of the study area ranges from subtropical to tropical, with mild to warm winters and hot summers (McNab et al. 2007). Mean annual air temperatures range from 17º to 24ºC and mean annual precipitation from 1,100 to 1,700 mm, with both increasing southward. Precipitation is evenly distributed through the year, though summer droughts are not uncommon, especially in the more northern and eastern sections of our study area.

Our shorthand name for the region (the "Gulf States") belies considerable geologic and soils variation across the study area. Even within the extensive areas of coastal plain, soil textures range from clay to coarse sand, and pH ranges from acidic to alkaline. In addition to coastal plain marine deposits, areas of alluvium and loess are also extensive, primarily along major river valleys and in the more western and interior portions of the study area. In the far west and interior, geology is dominated by sedimentary bedrock (including coal) and weathered sedimentary residuum. The

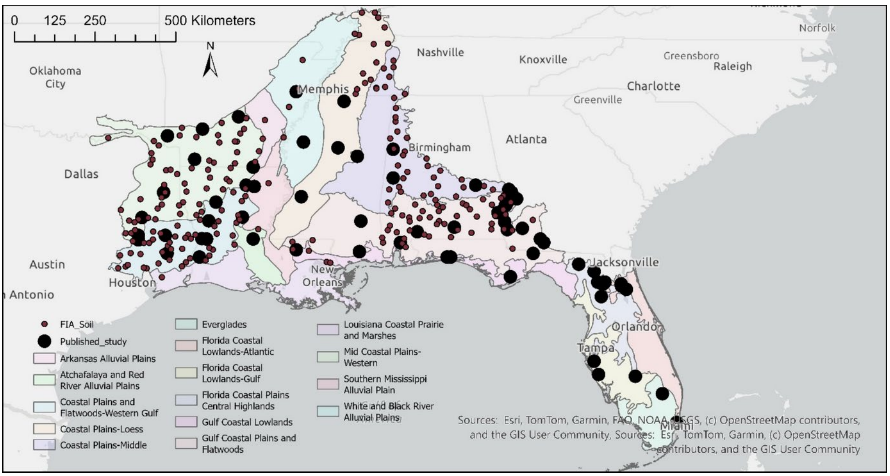

Fig. 1 Map of the study area. Shaded polygons are ECOMAP sections, large black points are locations of studies included in the meta-analysis, and small red points are FIA plots with O horizon data (FIA plots for biomass are two orders of magnitude more abundant across the region)

most consistent aspect of ecoregional physiography is its generally low relief, with plains being by far the most common landforms, followed by extensive riparian bottomlands, and a lesser extent of folded, faulted, or hilly terrain. Elevations range from sea level to approximately 225 m above sea level in the most interior portions of the study area in Georgia and Alabama (McNab et al. 2007). The region's diverse parent materials equate to diverse soils, which include highly weathered, strongly acid Ultisols formed in a wide range of parent materials, acid to alkaline Alfisols and Vertisols formed in clayey residual, alluvial or aeolian materials, very strongly acid Spodosols formed in coarse marine deposits, and occasional Inceptisols or Histosols formed in young parent materials or Holocene organic deposits, respectively (Stone et all 1993; Stiles et al. 2003; Shaw et al. 2010; Ahr et al. 2013).

The study area is approximately 60% forested, though land cover has become more dynamic in recent decades as urbanization has expanded into increasingly fragmented agricultural and forested landscapes (Griffith et al. 2003; Pinder et al. 1999; Nelson et al. 2020; Oswalt et al. 2017; Tatina et al. 2024; Woodbury et al. 2006). Of the forested lands, pine-dominated (*Pinus* spp.) systems are most extensive (especially planted stands), followed by oak-dominated (*Quercus* spp.) forests (oak with mixtures of hickory [*Carya* spp.] and/or pine), and bottomland hardwoods (including tupelo [*Nyssa* spp.], cypress [*Taxodium* spp.], and other wet-mesic to hydrophytic taxa (McNab et al. 2007). The region is a major producer and exporter of forest products ranging from bioenergy pellets to engineered wood products to lumber, largely driven by intensive management of southern pine (Dale et al. 2017; Jokela et al. 2010; Wear and Coulston 2025). Overall, southern pine forests, many of which are planted on former croplands, are typically managed using mechanical site preparation, artificial regeneration, fertilization, temporary control of competing vegetation with selective herbicides, and 1–2 thinnings before final harvest around 30 years (Fox et al. 2007). There is a wide range of specific silvicultural and operational tactics that fit into this overall management system, which frequently depend upon ownership, but in general it is more intensive than the management systems used in hardwood-dominated forests (Susaeta et al. 2023). Bottomland hardwoods are generally in settings with periodic to annual flooding regimes, with saturated soils, and are thus largely either unmanaged (e.g., riparian forests adjacent to pine upland forests), or managed with even-aged approaches that involve infrequent harvest entries (Kellison and Young 1997; King and Keim 2019). Upland oak-dominated forests include a wide range of accessory taxa and physiographic settings; overall, their management is less intensive than southern pine but more so than bottomland hardwoods. In oak-dominated systems, harvest entries are more frequent and involve crop tree or regeneration-focused treatments such as thinning, multi-stage shelterwood, and improvement cutting (Hicks et al. 2004).

Meta-Analysis

We used a meta-analysis approach detailed in previous ecoregional assessments, most recently Nave et al. (2025), to quantify the overall effects of various forest land uses (silvopasture, military training), land use changes (deforestation, reforestation, mineland reclamation, forested wetland creation) and forest management activities (prescribed fire, harvest, mechanical site preparation, non-harvest stand management) on soil C stocks. To the extent the data would allow, we then used meta-analysis to probe more nuanced effects of several specific forest management activities on soil and aboveground biomass C stocks.

This meta-analysis was based on data we synthesized from 48 papers, published between 2002 and 2022, identified through an extensive literature review (see Supplementary Information References). We determined each publication's suitability for inclusion using predetermined requirements. To be included, each paper conducted within the study area had to provide: 1) measured (not modeled) soil C data (concentration, stock, or horizon thickness [if for an O horizon]) for at least one relevant treatment and a paired control; 2) adequate metadata to constrain locations and use as potential predictor variables; 3) response data not included in previous publications. When available in these papers, we also extracted data on soil bulk density and aboveground live tree biomass as variables of interest, noting here that this opportunistic approach likely resulted in fewer publications on these variables than if we had designed our searching strategy to target them. In some cases, required information was not reported in the paper, but was provided in the online supporting information.

We extracted control and treatment soil C (or bulk density or tree biomass) values from each paper and used these to calculate effect sizes (as the *ln*-transformed response ratio R). As in our other published meta-analyses, most papers reported on multiple different conditions, and therefore were able to provide multiple response ratios, which across all papers spanned a wide range of forest types, soil depths, harvest practices, time since treatments, and other sources of variation. The purpose of meta-analysis was then to test whether these potential sources of variation significantly influenced effects of management on C stocks or bulk density. Response ratios also varied in terms of how their control conditions were defined; we detail those here to aid in assessing our statistical design and inferential scope. Prescribed fire papers were straightforward, being comparisons of paired burned vs. unburned stands. Forest harvesting papers were similarly straightforward (harvested vs. unharvested stands), while non-harvest stand management practices took a variety of forms. For example, based on the available data we were able to calculate response ratios for fertilization by comparing paired fertilized and non-fertilized stands at mid-rotation, and by comparing paired stands that had both been recently harvested, but only one of which was fertilized. In both cases, response ratios provided an estimate of the effect of fertilization, whether independent of or interacting with harvesting, and our approach to extracting and coding predictor variables for every response ratio allowed us to subsequently test whether the effect of fertilization was mediated by harvest. Land use change papers included studies of reforestation and deforestation. For agricultural reforestation papers, we accepted either recently abandoned fields (e.g., in oldfield succession) or active fields as controls, and their paired reforesting stands as treatment observations. In agricultural deforestation papers, we compared soil C stocks from never-farmed forests (as controls) to adjacent agricultural sites (as treatments). In mineland reforestation papers, treatment sites were those with trees present that had been planted at least 5 years prior, and control sites were those that had been mined and reclaimed very recently (< 5 yr) or not at all. Chronosequences pose a unique case; in our present analysis, most chronosequence papers involved forested wetland creation or mine reclamation with trees, and for these, we used the youngest stand age in the chronosequence as the control condition. In the case of one chronosequence paper, involving forest harvesting, we used the oldest stand age in the chronosequence as the control and compared the younger (harvested) stands to as the treatment stands.

We used MetaWin 3.0.15 (Rosenberg 2024) to conduct fixed-effects, categorical and continuous meta-analyses (Hedges et al. 1999; Hedges and Olkin 1985), with bootstrapped confidence intervals determined using the Student's t distribution (Adams et al. 1997; Dixon 1993). We chose to conduct an unweighted meta-analysis because many studies did not adequately report their sample sizes or variance statistics, which are needed for a weighted meta-analysis.

In this meta-analysis, soil organic carbon (SOC) stock (Mg C ha^{-1}) was our response variable of interest. When data were not reported in those units we converted them, as needed, using the same approaches as our other assessments (most recently summarized in Nave et al. 2025). Briefly, for studies reporting soil organic materials as the concentration of soil organic matter (%SOM, derived from loss on ignition; 48 of 915 response ratios), we converted %SOM to %SOC, assuming a fixed factor of 0.5. For papers that reported soil C as %SOC, we derived a relationship between %SOC and soil bulk density (Db) for the 98 observations that reported both variables and used the resulting model to predict Db from %SOC for papers only reporting the latter (%SOC; 71 of 915 response ratios). This equation took the form:

$$Db\ (g\ cm-3) = -0.1877 + (1.6424 * 10.3504)/(10.3504 + \%C) \quad (1)$$

With $P < 0.001$, $r^2 = 0.67$, and a standard error of the estimate of 0.24. We then computed SOC stock (Mg C ha^{-1}) as the product of the reported %SOC, the predicted Db, and the reported sampling interval depth or horizon thickness. We did not attempt to change reporting units or estimate SOC stocks for the 24 observations that reported soil organic contents as horizon thickness, all of which were O horizons.

To evaluate factors that may influence treatment effects on soil C stocks, we extracted a range of ecological, geographic, climatic, experimental, and methodological predictor variables from each paper. When needed, we located missing study site information in other publications from the same sites, or using geocoordinates reported in the papers, or accessed information about the soil series reported from those study sites, via the USDA Natural Resources Conservation Service Official Soil Series Descriptions (https://soilseries.sc.egov.usda.gov/osdname.aspx).

As in prior published assessments, we used meta-analysis to identify statistically significant predictors of variation in management effects on soil C stocks (or Db or aboveground biomass), which is done by parsing variation into within-group (Q_w) and between-group heterogeneity (Q_b) and inspecting corresponding P values (Higgins and Thompson 2002). Grouping variables that have large Q_b

relative to Q_w are statistically significant ($P < 0.05$) and explain a larger share of total heterogeneity among all studies (Q_t). The same is true for continuous meta-analysis (i.e., meta-regression), in which the variance explained by a predictor variable is denoted as Q_m (model-explained heterogeneity). However, for categorical meta-analysis, P values represent only one way to assess the significance of results. In this study, in addition to identifying statistically significant predictors based on their P values, we were interested in identifying groups of response ratios that did not have predictors with statistically significant P values, but which were significantly different from zero percent change based on their bootstrapped confidence intervals not overlapping zero. In all formal meta-analyses, we imposed a small sample size criterion, electing to exclude from discussions of statistical or practical significance any group that showed statistical significance (or was significantly different from zero percent change) but had fewer than $k = 5$ response ratios.

Forest Inventory and Analysis Data

We complemented our meta-analysis with data from the FIA program, an equal-probability sample of all forest lands in the conterminous US. Across the conterminous US, there is one permanent plot on approximately every 2,400 ha, randomly located within a hexagonal sampling frame (McRoberts et al. 2005). All FIA plots with at least one forest condition are measured every 5–7 years in the eastern US. Soils are sampled from a subset of these plots, according to a protocol in which organic horizons are removed and mineral soils are then sampled in depth increments of 0–10 and 10–20 cm. The FIA design ensures no systematic bias regarding location, ownership, composition, soil, physiographic factors, or other characteristics. We obtained data for this analysis from a July 2022 query of the FIA Database (FIADB) for records of aboveground biomass, organic horizon, 0–10 cm and 10–20 cm mineral soil C stocks (all in Mg C ha^{-1}). To ensure data consistency, we constrained the query to plots that were at least 75% under the same condition, excluding plots with sharp boundaries with different stand age, slope, wetness, etc., in which such variation might misrepresent conditions at the actual location of soil sampling. Moreover, we only used the most recent observation available in FIADB (USDA Forest Service 2022) of each FIA plot, and only plots observed since 2000. We used GIS lookups of FIA plot locations to obtain the following categorical attributes: mean annual temperature (MAT) and mean annual precipitation (MAP; PRISM 2015), surface geology (Soller et al. 2012), and soil taxonomic order (Soil Survey Staff 2022). Additional plot-level variables recorded by FIA included elevation, slope gradient and aspect, physiographic position, forest type, aboveground biomass, and disturbance and treatment histories. Our study reports results from $n = 183$, $n = 220$, and $n = 13{,}007$ plots for mineral soils, O horizons, and aboveground biomass, respectively, in which we used FIA data in specific analyses to contextualize and evaluate our meta-analysis results. We note here that our FIA data analyses were intended to be as similar in scope and granularity to the meta-analysis as possible, rather than an attempt to parse meta-analytic trends into greater detail, which may be ill-advised given the observational design and limited replication of FIA soils data.

To assess temporal trajectories of O horizon C stocks in (1) burned [prescribed fire] vs. unburned and (2) harvested vs. unharvested forests with FIA data, we first log-transformed C stocks to normalize their distribution and then identified the subset of plots with an observation of past prescribed fire or harvest, respectively. We then compared these plots statistically to a subset of control plots that matched the burned (or harvested) observations in terms of the following categorical variables: ecological subsection (ecological section in the case of harvested plots), surface geology, soil order, physiographic class code, forest type group, and site class code. We also ensured that, for each comparison, control plots fell within the same range of values as the burned (or harvested) plots for the following continuous variables: elevation, slope gradient, mean annual temperature, mean annual precipitation, live tree stocking percentage, and stand age. We introduced these constraints to ensure that plots being compared in their fire or harvest history were reasonably similar in physiographic and biotic parameters, increasing our confidence that any C stock differences between them were due to their fire or harvest history. The result of these exercises in each case (fire and harvest) was a well balanced and regionally widespread number of plots for each comparison, with considerably more plots for the harvest comparison ($n = 124$) than the fire comparison ($n = 13$). We then fit separate regression models to the stand age vs. O horizon C stock relationships for each of these four conditions (burned, unburned, harvested, unharvested), and used P and r^2 values as the criteria for selecting the best fit model.

We also examined temporal trajectories in O horizon, mineral soil, and aboveground biomass C stocks using data from FIADB, agnostic of their history of harvest, fire and other disturbances. First, we used FIA *fortypgrp* codes to group plots into three broad forest types: loblolly and longleaf pine (*P. taeda* and *P. palustris*; codes 140 and 160), bottomland hardwoods (codes 600 and 700), and oak-dominated (codes 400 and 500). We excluded the small number of plots from other forest type groups. We tested whether C stocks varied as a function of stand age and forest type group using multiple linear regression, with stand age as a continuous variable and forest type group coded as a categorical variable relative to the oak-dominated group. In these regressions, we eliminated a small number of observations $> 3\sigma$ from the mean in each variable (O horizons, biomass, and mineral soils) to minimize leverarging by highly non-representative conditions, and ln-transformed the data to normalize them. To aid in representing temporal trends graphically, we binned observations into five age classes with relatively balanced sample sizes: < 10 years, 11–20 years, 21–30 years, 31–50 years, and > 50 years.

We analyzed and plotted FIA data using Minitab Version 21 and SigmaPlot Version 14. In all cases, we used data transformations to normalize data distributions and equalize variances and set $P < 0.05$ as the threshold for accepting test results as statistically significant.

Results

Main Effects

Meta-analysis of published papers indicated that different forested land uses, land use changes, and management activities had significantly different effects on soil

C stocks (Fig. 2). Among land uses that maintained forest as forest, prescribed fire was the only action associated with a significant change (a decline in soil C stock); harvest, mechanical site preparation, and non-harvest stand management activities (fertilization, competing vegetation control) showed no overall significant effects on soil C stocks. In terms of land use changes, native forest conversion to planted forest, pasture, or developed land showed no significant effects on soil C stocks, but forest conversion to cultivated agriculture was associated with soil C stock declines. Conversely, reforestation after agriculture, on reclaimed mine lands, or on areas formerly deforested by military training activities was associated with significant increases in soil C stocks. Other forested land uses sufficiently reported in the literature included silvopasture, which was associated with no change in soil C stocks, forest use for military training, which was associated with significant soil C declines, and woody wetland restoration, which was associated with significant soil C gains.

Fire and Forest Management Effects in Focus

Prescribed fire had different effects on C stocks in different portions of the soil profile, with O horizons declining significantly and mineral soils showing no overall change (meta-analysis, $P<0.05$; Fig. 3). Among O horizons, studies reporting their results as horizon thickness tended to show larger declines than those reporting their results in terms of C stocks, but the difference was not statistically significant.

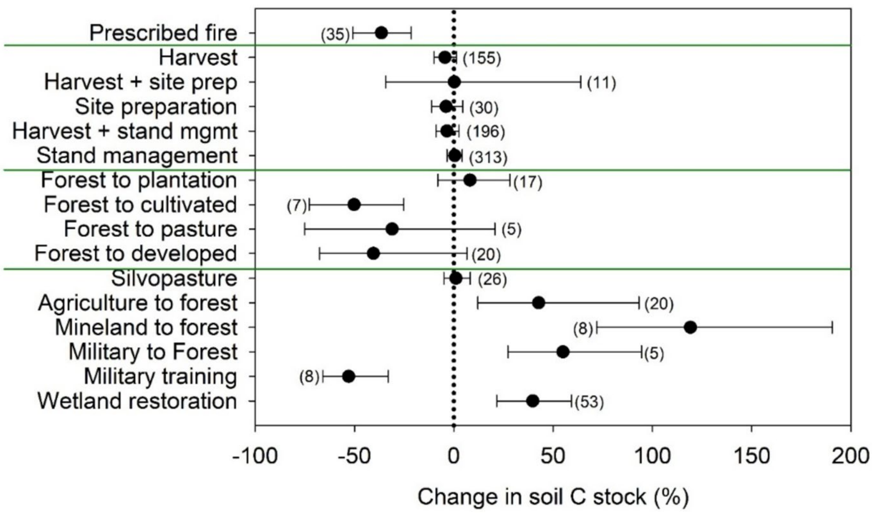

Fig. 2 Main effects of land use change and management activities on soil C stocks, according to meta-analysis. Points are mean effect sizes and error bars are bootstrapped 95% confidence intervals, with the number of response ratios in parentheses. Treatments overlapping the dotted vertical line show no significant change in soil C stocks. Horizontal green lines group similar treatments into categories of fire (top), forest management activities (second from top), land use changes from forest to more intensive uses (third from top), and intensive uses to forest (bottom)

Meta-analysis showed that forest harvesting and non-harvest stand management activities (fertilization, competing vegetation control) had different effects on different ecosystem C pools, which depended on whether the activities were performed independently or in combination (Fig. 4). Harvesting trees significantly decreased C stocks in aboveground live biomass but had no effects on O horizon or mineral soil C stocks, whether or not stand management activities were performed in conjunction with harvesting. In contrast, stand management activities performed independent of harvest (e.g., during the rotation) were associated with increased C stocks in aboveground live trees and O horizons, and declines in mineral soil C stocks.

A closer look at specific harvest and stand management activities revealed a range of significant effects on different portions of the soil profile. When applied independent of harvest, fertilization increased O horizon C stocks, whether or not competing vegetation control was also applied (Fig. 5a). When applied in conjunction with harvest, neither of these stand management practices had significant effects on O horizon C stocks (Fig. 5b). In mineral soils, fertilization had no effect by itself, but counteracted a negative effect of competing vegetation control on soil C stocks (Fig. 6). This pattern was the same whether these practices were applied in conjunction with harvest or independently.

All reported forest management activities, including harvest, mechanical site preparation, and non-harvest stand management significantly increased mineral soil bulk density (Fig. 7). The magnitude of this increase did not vary among different types of management activities.

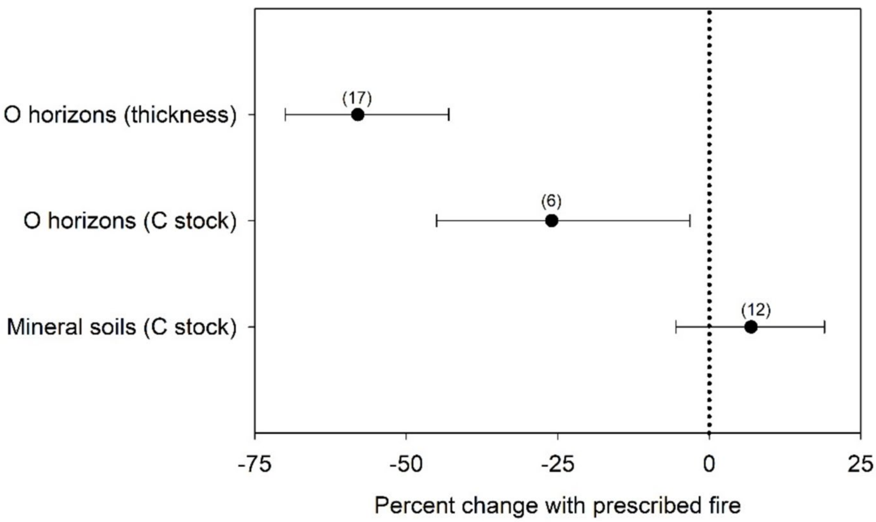

Fig. 3 Prescribed fire effects on soils as a function of soil horizon and reporting units, according to meta-analysis. For organic horizons, studies reporting in units of thickness (cm) and C stock (Mg C ha^{-1}) are shown separately; for mineral soils, all studies are reported in units of C stock. Points are mean effect sizes, error bars are bootstrapped 95% confidence intervals, with the number of response ratios in parentheses. Treatments overlapping the dotted vertical line show no significant effects of fire

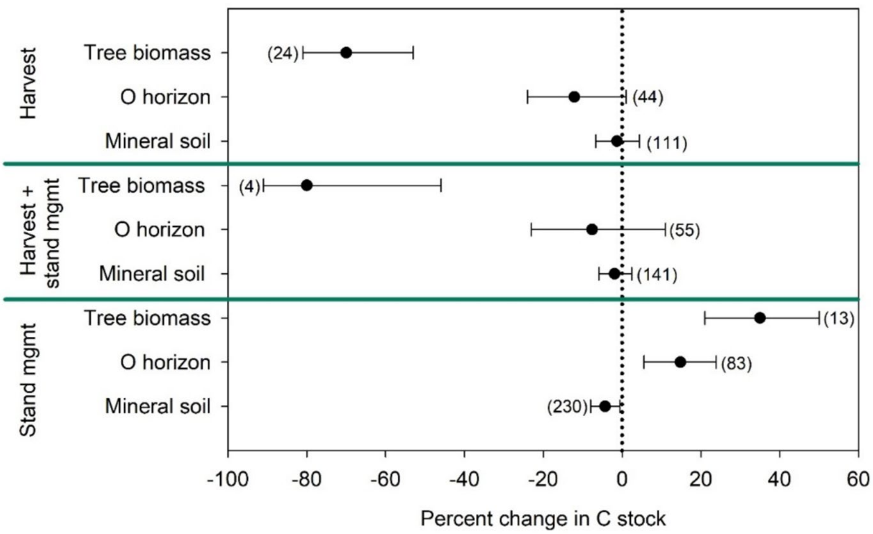

Fig. 4 Effects of harvest (top) and non-harvest stand management activities (bottom; fertilization and competing vegetation control) alone, or in combination (middle), on C stocks in aboveground live tree biomass, organic horizons, and mineral soils, according to meta-analysis. Points are mean effect sizes and error bars are bootstrapped 95% confidence intervals, with the number of response ratios in parentheses. Treatments overlapping the dotted vertical line show no significant change in C stocks

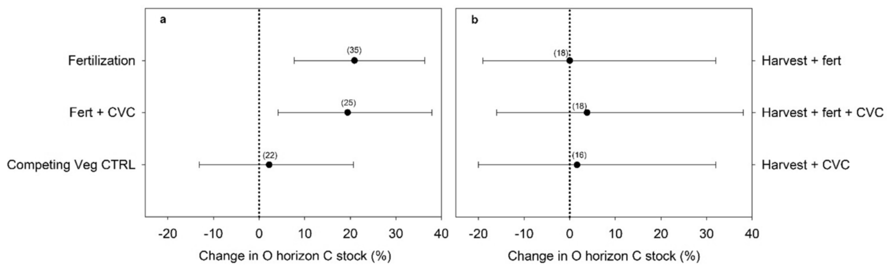

Fig. 5 Effects of stand management activities on O horizon C stocks, independently (**a**) or in conjunction with harvest (**b**). Activities include fertilization (fert) and competing vegetation control (CVC). Points are mean effect sizes and error bars are bootstrapped 95% confidence intervals, with the number of response ratios in parentheses. Treatments overlapping the dotted vertical line show no significant change in C stocks

Evaluation of Meta-Analysis Trends with FIA Data

Comparisons of O horizon C stocks in burned vs. unburned and harvested vs. unharvested forests revealed that these activities were associated with different trajectories of C stock change vs. stand age (Fig. 8). The limited number of burned plots showed an asymptotic increase in O horizon C stocks as a function of stand age, leveling

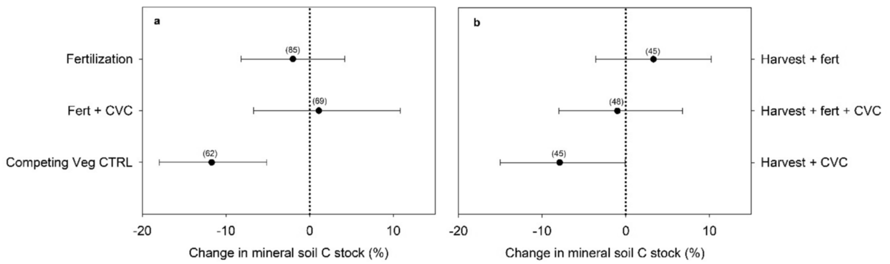

Fig. 6 Effects of stand management activities on mineral soil C stocks, independently (**a**) or in conjunction with harvest (**b**). Activities include fertilization (fert) and competing vegetation control (CVC). Points are mean effect sizes and error bars are bootstrapped 95% confidence intervals, with the number of response ratios in parentheses. Treatments overlapping the dotted vertical line show no significant change in C stocks

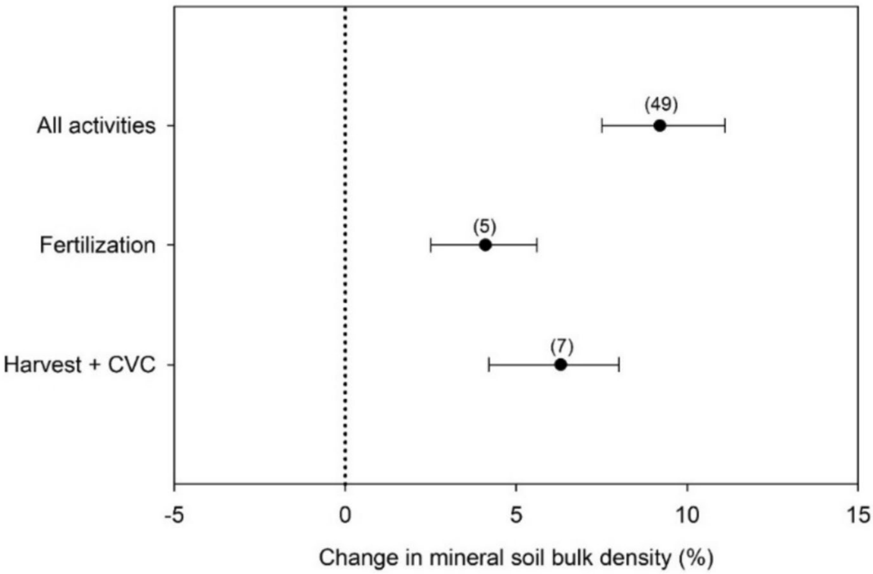

Fig. 7 Effects of forest management activities overall (top, including all harvest, site preparation, and stand management), fertilization only (middle), or harvest and competing vegetation control (CVC; bottom) on mineral soil bulk density. Points are mean effect sizes and error bars are bootstrapped 95% confidence intervals, with the number of response ratios in parentheses

off at slightly more than half the C density of unburned plots, despite representing a longer continuum of stand ages. Unburned plots tended to be in younger forests, but showed rapid, linear O horizon C gains. The more regionally abundant data on harvesting showed an asymptotic increase in O horizon C stocks across stand ages after harvest, while unharvested stands showed no significant relationship between stand age and O horizon C stocks. According to best-fit models describing O horizon C stocks in

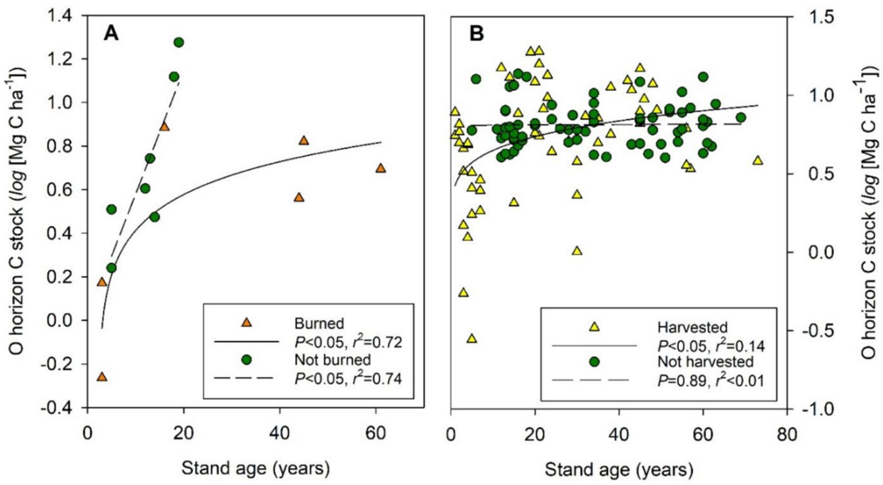

Fig. 8 Organic horizon C stocks as a function of stand age for burned vs. unburned forests (panel **A**) and harvested vs. unharvested forests (panel **B**), according to FIA data. Points plotted are individual FIA plot observations

harvested and unharvested forests, the point at which harvested stands attained unharvested levels was at a stand age of *ca.* 30 years.

FIA data showed that across the study region, O horizon and aboveground biomass C stocks increased with stand age, at rates that depended on forest type (Fig. 9). Organic horizon C stock increases with stand age were significantly larger in pine-dominated forest types than oak-dominated forest types, which were in turn significantly larger than bottomland hardwoods (Table S1). However, the proportion of unexplained variance in the multivariate model was quite large, ($R^2=0.08$). Aboveground biomass C stocks increased at similar rates with stand age for oak-dominated and bottomland hardwood forests, while pine-dominated forest types showed significantly larger increases with stand age. Variance in this relationship was lower than for O horizons but still considerable (Table S2; $R^2=0.35$). Mineral soil C stocks to 20 cm depth did not vary with stand age for any of the forest types, nor did they differ by forest type.

Discussion

Land Use Change and Soil C Stocks

In the Gulf States, land use changes generally tend to have larger effects on soil C stocks than management activities conducted in forested lands remaining forested, which showed no significant effects at a regional scale. This overall trend is consistent with results for other ecoregions of the US, summarized most recently in Nave et al. (2025), though the specific land use changes that affect soil C stocks are not the same in all regions. The decrease in soil C stocks that accompanies deforestation for agriculture in the Gulf States is similar in magnitude to other southern and

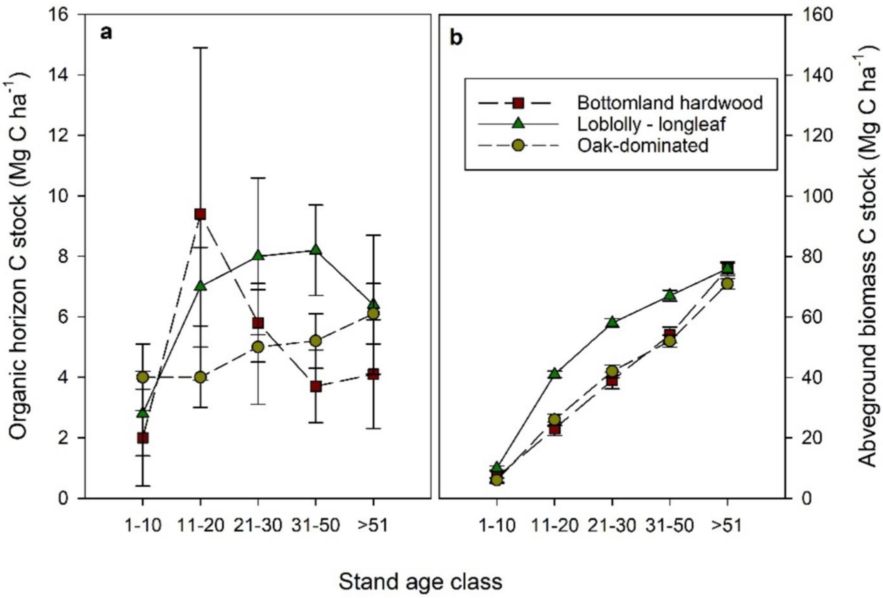

Fig. 9 Carbon stocks in O horizons (**a**) and aboveground live biomass (**b**) as a function of forest type and stand age class, based on FIA data. Points plotted are means of individual FIA plots; error bars represent 95% confidence intervals. Note differing y-axis ranges

coastal regions of the US, but this type of land use change is mostly a historical phenomenon (Harden et al. 1999; King and Keeland 1999; Nave et al. 2019; 2022a; Schoenholtz et al. 2001; Upadhaya and Dwivedi 2019). In recent decades, deforestation has largely been driven by urban and residential development, which does not appear to affect soil C, but does negatively affect other ecosystem services and C pools (Li et al. 2022; Riiters et al. 2020; Sun et al. 2005; Woodbury et al. 2006).

Reforestation of former agricultural lands is extensive throughout the lower Mississippi valley and Gulf States, both naturally following agricultural abandonment and intentionally by planting trees for southern pine production forestry (Berkowitz 2013; Gordon and Barton 2015). In combination with this large extent, the significant soil C stock increases with this land use change imply a considerable C sink where forests are established or continuing to recover on former croplands (Cho et al. 2025; Nave et al. 2018; Switzer et al. 1979; Zhang et al. 2023). This C sink is further enhanced when considering equal or larger rates of increase in aboveground biomass C compared to soil C, especially for the highly productive southern pine forests that represent a large share of the reforestation in the southeastern US (Fox et al. 2007; Shoch et al. 2009; Sun et al. 2015).

Woody wetland restoration also appears to offer opportunities for soil C sequestration in the Gulf States. Our meta-analysis included only three published studies of this land use change which, despite very different starting conditions, showed consistently large soil C stock increases (Berkowitz 2013; Berkowitz et al. 2013; Oslund et al. 2020). While three papers are not sufficient to quantify soil C recovery rates, the consistency of soil C gains implies that coastal wetland and bottomland forest restoration present opportunities to contribute to a regionally important C sink (Krauss et al. 2017; McCarthy et al. 2022; Sapotka and White 2020). This argues for further studies of a wider range of sites, starting conditions, and restoration approaches to constrain C gains and assist in restoration prioritization (Osland et al. 2022). If woody wetland restoration is to provide durable C sequestration and climate change mitigation, then additional research will also be needed to quantify the effects of increased climatic disturbances, most notably hurricances, on C stocks and fluxes (Griffiths and Mitsch 2021; Jiang and Middleton 2011).

This is the first ecoregion we have assessed with a sufficient number of studies to quantify soil C consequences of military activities, including training under forest cover, and reforestation after training and testing activities previously eliminated the forest cover. Our meta-analysis results showing significant and fairly consistent effects of these changes should be considered alongside several caveats. First, like wetland restoration, the number of studies reporting on these effects was limited, and even more geographically restricted; the three papers on the topic were all focused on activities at Fort Benning, Georgia (Jolley et al 2010; Maloney et al. 2008; Silveira et al. 2010). This gives rise to the second caveat: while most large military installations have long supported ecosystem management and restoration activities, the nature of these activities (and the training and testing activities that are their primary objective) differ substantially among installations (Christensen et al. 2021; Goodman 1996; Lavoie et al. 2014). Given the sensitivities of collecting data at these facilities, and restrictions on where research can be done within them, it is likely that our meta-analysis represents a subset of the broader range of effects that military activities have on soil C stocks at Gulf States installations. It would thus be inappropriate to extend the inferences from our meta-analysis to other installations, where activities (and baseline ecosystem properties) may be quite different.

In Focus: Forest Management Effects on C Stocks

Overall, our meta-analysis indicates that forest management has a limited capacity to affect soil C stocks, as compared to land use change. However, this aggregate regional result obscures the real and significant effects (positive and negative) of specific activities, and some evidence for mediation of these effects by spatially variable site factors. Results based on FIA data lend confidence to several of these findings.

First, both data sources (meta-analysis and FIA plots) indicate that prescribed fire significantly decreases O horizon C stocks. The typical reduction based on these two data sources is roughly half the starting O horizon C stock, which is similar to effects in other ecoregions, as summarized in Nave et al. (2025). However, variability in the meta-analytic effect size that we report here, coupled with the diversity of stand conditions and burn prescriptions represented in our data sources, underscores that prescribed fire can be tailored to have a range of effects on O horizon C stocks (Kreye et al. 2014; 2017). This becomes more important when considering that managers typically do not cite C as the reason for implementing prescribed fire, which is more often used to meet other goals and objectives. These can include biodiversity or wildlife habitat, fuel or residue management, seed bed preparation, and more (Han et al. 2020; Kilburg et al. 2014; Kominoski et al. 2022; Lavoie et al. 2010; 2012; Martin et al. 2015). Depending on which goals and objectives are the priority for a given ecosystem, burn intervals mostly range from 1–5 years in the region, with desired fire effects ranging from partial fuel reduction and no effects on woody vegetation, to substantial understory mortality and mineral soil exposure (Kupfer et al. 2022; Mitchell et al. 2006; Whelan et al. 2018). These short return intervals are needed given the high rates of aboveground production and fuel accumulation in southern pine systems. Thus, the studies we synthesized for meta-analysis largely represent the temporary effects of fire as a frequently used management tool, rather than a one-time disturbance with persistent effects.

Compared to the high-confidence, significant effects of prescribed fire on O horizons, we find mixed evidence for an overall harvest effect. This evidence comes in the form of a statistically significant decline in the observational FIA data, alongside a directionally negative but highly variable and statistically non-significant meta-analytic effect size. Given this mixed evidence, we discuss this potential trend here in the context of risk tolerance in C management. In settings where loss of O horizon C stocks would be unacceptable, or where site factors point to the O horizon as at greater risk or particularly important from a soil productivity standpoint, harvest prescriptions or operations can be tailored to minimize the potential for negative effects. For example, where coarse soil texture or low organic matter limit moisture or nutrient availability, or the potential for runoff is high, careful retention and arrangement of live trees and residues can minimize the potential for soil warming, altered microbial community structure, erosion, and losses of surface organic matter (Carter et al. 2002; Gonzalez-Benecke et al. 2014; Mushinski et al. 2017; 2018; Zerpa et al. 2010). Making these decisions in risk-averse settings may be facilitated by existing soil protection guidelines related to canopy retention, streamside management zones, and layout of harvest and processing operations (Cristan et al. 2015; Hawks et al. 2022; NCASI 2009; Sanders and McBroom 2013). Further, we point out that even when O horizon C losses do occur, they represent a small share of the profile total C stock (on the order of 10% among the studies we included in our meta-analysis), and they appear to recover over the timescale of a typical pine rotation (~30 years based on FIA data).

Our results show that harvest and stand management are largely having the intended and logically expected results on ecosystem C pools. Harvesting removes aboveground biomass C and may modestly diminish O horizon C stocks; stand management activities intended to increase regeneration and tree growth increase C stocks in both pools, but may come at the expense of mineral soil C. For a closer look, we grouped these stand management activities into two categories: (1) competing vegetation control and (2) fertilization. These activities encompass many specific prescriptions and actions, such as the products applied, the timing, frequency and amount of application (individual and cumulative), and the type of equipment used to apply them, which can have nuanced and site-dependent effects on C stocks (Rifai et al. 2010; Tumushine et al. 2019; Vogel et al. 2011, 2021). Detailing those effects is beyond the scope of this paper, which instead emphasizes the overall regional consistency of these broad practices. In this regard, several findings related to belowground pools rise to the top as most relevant to operational forestry in the Gulf States. First, it appears that fertilization and competing vegetation control have larger positive effects on O horizon C stocks when conducted independently of (e.g., during a rotation), vs. in conjunction with harvest (e.g., immediately afterwards). Second, while the decline in mineral soil C stocks with competing vegetation control is a decline in a C pool that is significant at the ecosystem level, the offsetting effect of fertilization indicates that these two practices in conjunction can maintain soil C stocks while also supporting the management goal that has motivated them for decades: to successfully establish and regrow trees over multiple rotations (Fox et al. 2007).

Overall, our findings reinforce others suggesting that production forestry and C management are mutually compatible in the Gulf States (Loehle et al. 2009; Shephard et al. 2022). However, our Db results indicate a notable negative effect of management: soil compaction. This is the first of our ecoregional assessments to detect a significant overall effect of management on Db. That this effect was a universal increase, with no apparent mediation by any tested site or practice factors, makes it a high-confidence regional generalization. On its own, a Db increase of the magnitude we detected (5–10%) may be insufficient to affect soil processes. However, increasing Db diminishes root development, decreases aeration and infiltration, and increases runoff, and thus has the potential to affect productivity and ecosystem services in these managed forests (Miwa et al. 2004; Scott et al. 2014; Foote et al. 2015). Given that production forestry in the Gulf Coast consists of repeated rotations, there is a real potential for cumulative impacts, even if the Db increases we report here are not of immediate concern. Where these increases are judged as a concern, mitigation and amelioration measures may include aerial (rather than ground-based) application of herbicides or fertilizers, or use of deep tillage during site preparation. These practices, which are established within the southern pine management toolbox, may work in concert with natural recovery mechanisms such as shrinking and swelling of clays, pedoturbation by soil fauna, and root dynamics, to restore more natural soil structure (Lang et al. 2016; NCASI 2004).

Regional relationships between stand age and C stocks based on FIA data provide useful context for the forest types and management systems that we have generalized here. From a soils perspective, the lack of a stand age function for mineral soils reaffirms the lack of an overall effect of management on the largest ecosystem C pool. In contrast, O horizons show temporal dynamics that vary among the three broad forest type groups we have defined here. However, the spatial distribution of forest types reflects bottom-up physiographic factors, and as such the stand age functions that we report also reflect the influence of these physiographic factors (Goebel et al. 2001; Van Kley and Turner 2009). For example, the faster rate of O horizon C accumulation in bottomland hardwoods than upland oak-dominated systems may be as much related to hydrology or soil texture as to differences in forest composition. As a result, where oak forests occur on bottomlands, their C accumulation rates may differ from regionally representative rates, and use of these age functions for predictive purposes should therefore be made with caution. Hardwood distinctions aside, the considerably larger rates of C accumulation in O horizons and aboveground biomass in pine systems sets them apart. We aggregated all pine forest types into one group, obscuring important interspecific variation in biology and management, yet it is still clear by comparison to the hardwoods that the southern pine systems are more effective C sinks over a multi-decadal timescale. In this regard, it is important to note that the actual C sink attributable to pine systems over a rotation is higher than represented in our results, which do not explicitly account for C removed in thinnings or final harvest. Much of the C removed during thinnings is used for energy generation, displacing fossil fuels, and a significant fraction of the C removed in harvested wood products remains there for decades into the subsequent rotation (Dwivedi et al. 2014; Gonzalez-Benecke et al. 2010; Johnsen et al. 2001; Puls et al. 2024; Smith et al. 2006). These substitution and storage terms increase the forest sector C sink beyond the already strong C uptake attributable to aboveground biomass regrowth, further tipping the greenhouse gas balance of southern pine forest management in the favor of climate change mitigation.

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Data Availability The data underlying this paper come from public databases (USDA Forest Service, Forest Inventory and Analysis Program) and previously published papers analyzed using meta-analysis. Synthesized versions of these previously published data resources will be made available upon requests of the corresponding author.

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Authors and Affiliations

Lucas E. Nave[1] · Kendall DeLyser[2] · Grant M. Domke[3] · Scott M. Holub[4] · Adrienne B. Keller[1] · Darren Miller[5] · Kevin A. Solarik[6] · Erik Schilling[5] · Jason Vogel[7] · Christopher W. Swanston[8]

✉ Lucas E. Nave
lenave@mtu.edu

1 College of Forest Resources and Environmental Science, Michigan Technological University, Houghton, MI 49931, USA

2 American Forests, Washington, D.C. 20005, USA

3 USDA Forest Service, Northern Research Station, St. Paul 55108, USA, MN

4 Weyerhaeuser Company, WA 98531 Centralia, US

5 National Council for Air and Stream Improvement, Inc. (NCASI), Cary, NC 27511, USA

6 National Council for Air and Stream Improvement, Inc. (NCASI), Montréal, Québec H3A 3H3, Canada

7 School of Forest, Fisheries, and Geomatics Science, University of Florida, Gainesville, FL 32611, USA

8 Save the Redwoods League, San Francisco, CA 94104, US