Geoderma Regional 40 (2025) e00930

Contents lists available at ScienceDirect

Geoderma Regional

journal homepage: www.elsevier.com/locate/geodrs

Land use change and forest management affect soil carbon stocks in the central hardwoods, U.S.

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ARTICLE INFO

Keywords:
Central hardwoods
Deforestation
Forest management
Mine reclamation
Prescribed fire
Reforestation

ABSTRACT

Most research addressing land use change and forest management effects on soil carbon (C) is conducted at large or localized scales, rather than intermediate scales where management is planned and implemented. We assessed effects of land use and forest management on soil C stocks, for the Central Hardwoods ecoregion of the U.S., using meta-analysis, soil survey and national forest inventory databases to examine baseline controls on soil C stocks and their responses to land use and forest management. Biotic and geologic factors drive baseline variation in soil C stocks across the ecoregion, with forest type and productivity being most important in surface horizons and parent material dominating at the whole profile level. Among forest management treatments, prescribed fire is most noteworthy, decreasing O horizons to an extent determined by place and practice (mean: −53 %). Coal mine reclamation is extensive in the region, and while there is no effect of forest vs. herbaceous reclamation, distinct overburden types have different effects on soil C stocks (mean: +183 %). Land use change effects on soil C are difficult to determine due to the preferential use of the most favorable soils for agriculture, the relegation of forests to the least productive soils, and the tendency for reforestation to occur on marginal soils. Overall, our results can help forest managers anticipate the C outcomes of typical burn prescriptions in this region of extensive prescribed fire, and help landowners and planners understand how parent material and soil properties influence soil C stocks under agriculture and mine reclamation.

1. Introduction

Soil carbon (C) is a key component of forests' ecological, biogeochemical, and hydrologic processes, underpinning numerous ecosystem processes and services (Vance, 2000; Nave et al., 2019a). Because of soils' importance and their critical role in climate change mitigation strategies, global interest in the effects of land use change and forest management on soil C has been growing for some time (Minasny et al., 2017; Harden et al., 2018).

The international perspective on soil C and climate change mitigation has led to an abundance of high-level reviews of land use change and forest management effects on soil C stocks (e.g., Post and Kwon, 2000; Certini, 2005; Jandl et al., 2007; Laganiere et al., 2010; Nave et al., 2010, 2011; Thiffault et al., 2011; Lorenz and Lal, 2014; James and Harrison, 2016; Smith et al., 2016; Dignac et al., 2017; Mayer et al., 2020). However, this review literature holds many generalizations that do not align with the findings of individual site-level studies. This disparity emphasizes the need for research on soil C management at intermediate (landscape to ecoregional) scales, which are often the focus of decision making by landowners, forest managers, and policymakers.

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https://doi.org/10.1016/j.geodrs.2025.e00930
Received 5 November 2024; Received in revised form 15 January 2025; Accepted 7 February 2025
Available online 8 February 2025

By linking the disparate large- and small-scale studies of land use, management, and soil C, intermediate analyses can advance global-scale progress on soil C management while remaining sensitive and responsive to more local human constraints, management systems, and ecological drivers.

As data from primary research on land use, forest management, and soil C have accumulated, intermediate-scale soil C management studies have increasingly become able to harness synthesis approaches that in the past were only useful at broad scales, where data were abundant. This utility stems partly from the flexibility of synthesis tools. For example, meta-analysis can estimate overall treatment effects across multiple studies while also identifying factors that drive variation in these effects (Hedges et al., 1999). However, all meta-analyses are constrained by the availability of primary studies; their strength in detecting trends across experimental sites is accompanied by an inability to address the diversity of conditions across intervening spaces or conclusively demonstrate mechanisms (Gurevitch et al., 2001). Drawing in more extensive observational data, such as from soil survey and forest inventory programs, can help mitigate these limitations. Observational datasets can lack experimental control compared to the studies synthesized for meta-analysis, may not have desired auxiliary variables, and introduce other sources of variation that can obscure or confound trends. Nonetheless, observational data enable inferences over intervening areas that have not been studied, and auxiliary variables can be obtained from other sources to evaluate and contextualize meta-analysis results (Fick et al., 2021). Combining meta-analysis with observational data has been useful for downscaling from broad patterns (e.g., Nave et al., 2010, 2018) to distinct ecoregions (Nave et al., 2019b, 2021, 2022a, 2022b, 2024), and holds promise for similar refinements in still more places.

The U.S. Central Hardwoods ecoregion (Fig. 1) is a nearly 600,000 sq. km area that is roughly three-fourths forested. The region is situated at the confluence of several geographic and geologic provinces that define both its natural character and the human endeavors that occur within it. Its oak-hickory, oak-pine, and mixed hardwood forests support extensive forest management, with short-return surface fires playing notable roles in both historical and contemporary times (Knapp et al., 2014; Luppold and Bumgardner, 2018; Abrams et al., 2022). Moving to the north and west, as the topography grows less rugged and the soils more productive, forestland gives way to agriculture (Fralish, 1994; Schaetzl et al., 2012). Throughout the region, coal deposits occur in various bedrock types, and the long history of mining has influenced the land and its forests in many ways (Greb et al., 2020; Hower et al., 2022). The resulting combination of forest, agricultural, and mining land uses in the Central Hardwoods, coupled with increasing interest in soil C management for climate mitigation, create the need for a focused assessment of soil C stocks in the region. The present synthesis, representing the sixth in a series of ecoregional assessments, is intended to provide that assessment by investigating baseline controls and land use and forest management effects on soil C stocks. As in our prior assessments for other ecoregions, our objectives are threefold: (1) identify the factors that drive baseline variation in soil C stocks; (2) constrain the direction, magnitude, and variability in land use and management effects on soil C stocks; (3) place land use and management effects in the

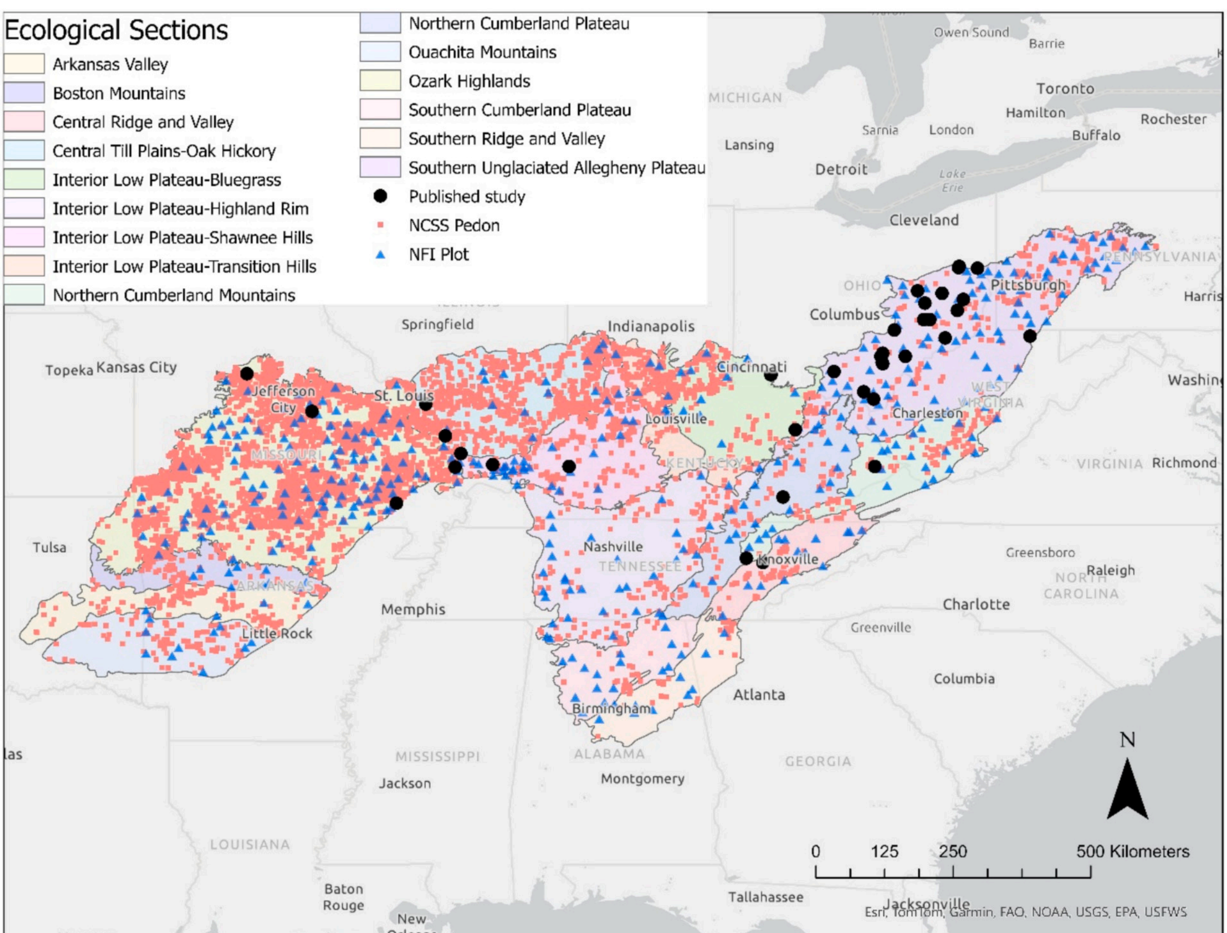

Fig. 1. Map of the study area. Shared polygons are ECOMAP sections, black points are locations of published studies included in the meta-analysis, pink squares are NCSS pedons, and blue triangles are NFI plots.

context of baseline drivers and variation to produce practical applications for managers.

2. Methods

2.1. Study area

Following previously published assessments (Nave et al., 2019b, 2021, 2022a, 2022b, 2024), we delineated the study area for this analysis according to the U.S. Department of Agriculture Forest Service (USDA FS) ECOMAP system (Cleland et al., 1997; McNab et al., 2007). Our focus was on the mostly forested, principally unglaciated Central Hardwoods region, as constrained by the boundaries of three previously published ecoregional assessments (Nave et al., 2019b, 2021, 2022b). We synthesized data from the 15 ECOMAP Ecological Sections comprising the study area, encompassing portions of the states of Pennsylvania, West Virginia, Ohio, Indiana, Illinois, Missouri, Oklahoma, Arkansas, Tennessee, Alabama, North Carolina, Kentucky, and Virginia (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 hot continental to subtropical, with cold to mild winters and warm to hot summers (Brandt et al., 2014; McNab et al., 2007). Mean annual air temperatures range from 7° to 17° and mean annual precipitation from 900 to 1700 mm. The northern and eastern portions of the study area are the coolest, precipitation generally increases to the south and east (and with elevation), and all areas have humid climates, with precipitation far exceeding potential evapotranspiration. Approximately three-fourths of the region is forested, with cover types ranging from maple-beech-birch in the northeast, to oak-hickory and oak-pine in the west and southwest. Overall, oak-hickory is the dominant forest type (Brandt et al., 2014). Elevations range from 60 m above sea level in the Mississippi River valley to >1100 m above sea level along the western flank of the Appalachian Mountains. The geology of the study area is varied. In northern and eastern portions, sandstone, shale, and siltstone dominate, with limited limestone and dolomite; to the south and west, these carbonate sedimentary rock types become more common. Limited areas are dominated by igneous rocks (portions of the Ozark Highlands), or have surface deposits of glacial drift (mostly till) or loess, in the northwestern portion of the study area (McNab et al., 2007; Nelson and Lumm, 1984). Across most of the study area, the topography is mature and dissected, especially in mountainous areas; areas of glacial deposits have lower relief and more recently developed drainage networks. The most extensive soil orders are Alfisols (Luvisols) and Ultisols (Acrisols), both formed in clay-rich parent materials, with Alfisols more common in areas of (calcareous) glacial till and carbonate residuum, and Ultisols more commonly forming in the residuum of non-calcareous sedimentary rock types. Mollisols (Phaeozems) are common, especially to the north and west and in alluvium and calcareous sedimentary residuum. Entisols (Regosols) and Inceptisols (Cambisols, Leptosols) are less extensive and occur in alluvium and colluvium, respectively. Further description of the study area's climate, vegetation, geology, and soils is provided by McNab et al. (2007).

2.2. Approach

We used synthesis methods detailed in previous ecoregional assessments (Nave et al., 2019b, 2021, 2022a, 2022b, 2024). Methods included: (1) effect size meta-analysis of data from published papers; (2) synthesis of soil survey pedon observations with GIS information; (3) analyses of National Forest Inventory (NFI) data from plots in which soils, biomass, and other ecosystem properties were measured. We employed these methods as follows. First, we analyzed soil survey and NFI data to identify baseline factors controlling variation in soil C stocks across the study area. Next, we used meta-analysis to quantify the effects of land use change, mine reclamation, and forest management on soil C stocks. To the extent published data would allow, we then used meta-analysis to investigate more detailed trends in several land use and forest management categories. Finally, we analyzed soil pedon and NFI data as two independent sources of information to contextualize and evaluate meta-analytic trends to the extent possible.

2.3. Meta-analysis

We synthesized data from 26 relevant papers, published between 2004 and 2021, identified through an extensive literature review (identified with a * in the Supporting Information References). As with our prior meta-analyses, we determined each publication's suitability for inclusion using predetermined requirements. More specifically, to be included, each paper falling within the study area had to provide: 1) control and treatment values for soil C (concentration, stock, or horizon thickness [if for an O horizon]) for at least one relevant treatment; 2) adequate metadata to constrain locations and use as potential predictor variables; 3) response data not included in previous studies. In some cases, required information was not reported in the paper, but was provided in the online supporting information.

From all 26 papers, we extracted control and treatment soil C values and used these to calculate effect sizes (as the ln-transformed response ratio *R*). As in our other published meta-analyses, these response ratios span a wide range of forest types, soil depths, amount of time elapsed since treatments, and other sources of variation. Response ratios also vary in terms of how their control conditions were defined. Studies of mine reclamation and land use change are noteworthy in this regard and require specific explanation. In mine reclamation papers, we used the most heavily mining-affected observation as the control condition. In most cases, this was the initial site in a chronosequence, e.g., a 2-year-old restored site, with treatment observations then reported along a multi-decadal chronosequence. In other cases, the control was a not-yet-reclaimed site, compared to adjacent reclaimed sites. Land use change papers included studies of reforestation and deforestation. For reforestation studies, we accepted either initial chronosequence stands (e.g., in oldfield succession) or active agricultural sites as controls, and their corresponding reforesting stands as treatment observations. In deforestation to agriculture papers, we compared soil C stocks from never-farmed forests (as controls) to adjacent agricultural sites (as treatments). In deforestation to mineland papers, control sites were undisturbed forests and treatment sites were those that had been mined and had been reclaimed very recently (<5 yr) or not at all.

We used MetaWin 3.0.15 (Rosenberg, 2024) to conduct fixed-effects, categorical and continuous meta-analyses (Hedges and Olkin, 1985; Hedges et al., 1999), with bootstrapped confidence intervals determined using the Student's t distribution (Dixon, 1993; Adams et al., 1997). We chose to conduct an unweighted meta-analysis because of the large proportion of studies that did not report the sample size and variance statistics needed for a weighted meta-analysis. We also did not assume that the data met the parametric assumptions of 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 (Nave et al., 2019b, 2021, 2022a, 2022b, 2024). For studies reporting soil organic materials as the concentration of soil organic matter (%SOM, derived from loss on ignition; 26 of 337 response ratios), we converted %SOM to %SOC, assuming a fixed factor of 0.5. For papers that reported soil C as %SOC, we derived land-use-specific relationships between %SOC and soil bulk density (Db) for the 176 observations that reported both, and used the resulting models to predict Db from %SOC for papers only reporting the latter (%SOC; 74 of 337 response ratios; Table 1). 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. In one paper (k = 6 response ratios) involving mine reclamation, the control soil C stock

Table 1
Linear relationships between soil C concentration and bulk density (Db) derived from published studies reporting both parameters. Linear models were derived separately for soils from forest management studies (Forest; harvesting or prescribed fire), land use change (LUC; reforestation or deforestation), and mine reclamation (Mining). For each model, the equation, standard error of the estimate, r^2, and P values are shown, with the number of observations and their ranges of values.

Category	Model	SE	r^2	P	n	Db range	%C range
Forest	Db = 1.511 - (0.139 * %C)	0.15	0.55	<0.001	51	0.91–1.71	0.20–3.9
LUC	Db = 1.681 - (0.203 * %C)	0.03	0.97	<0.001	22	0.95–1.61	0.28–3.4
Mining	Db = 1.762 - (0.187 * %C)	0.13	0.48	<0.001	103	1.11–1.99	0.43–3.4

value was 0; we set this equal to 0.01 Mg C ha^{-1} to compute a response ratio and include this study in the dataset.

To evaluate place-based and practice-specific 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 (USDA NRCS) online Official Soil Series Descriptions (https://soilseries.sc.egov.usda.gov/osdname.aspx, last accessed 8/15/2024). When key information was missing, but geocoordinates were provided, we used GIS lookups (section 2.4) to obtain the necessary attributes for each study.

As in prior published assessments, we used meta-analysis to identify statistically significant predictors of variation in soil C stock responses to management, 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 also concerned with identifying groups of response ratios that did not have predictors with statistically significant P values, but which were significantly different from 0 % change based on their bootstrapped confidence intervals not overlapping zero. In all meta-analyses, we imposed a small sample size criterion, electing a priori to exclude from discussions of statistical or practical significance any group that showed statistical significance (or was significantly different from 0 % change) but had fewer than k = 5 response ratios.

2.4. Synthesis of soil pedon and GIS data

We complemented meta-analysis, which tests for patterns among a limited number of intensively studied sites, with an extensive soil pedon dataset for our study area. This dataset consisted of geo-located soil pedons from the USDA NRCS, National Cooperative Soil Survey (NCSS) Database (July 2019 version), including latitude, longitude, soil taxonomy, and physical and chemical properties of individual soil horizons according to Schoeneberger et al. (2012) and Burt and Soil Survey Staff (2004). In addition to the data contained in the NCSS Database, we used GIS to extract the following attributes for each geo-located NCSS pedon: land cover from the most closely coincident version of the National Land Cover Dataset (NLCD; Jin et al., 2019; Homer et al., 2020; Dewitz and U.S. Geological Survey, 2021), aboveground biomass density from the National Biomass Carbon Dataset (Kellndorfer et al., 2007-2009, 2013), mean annual temperature (MAT) precipitation (MAP), and potential evapotranspiration (PET) for the 1981–2010 period (PRISM Climate Group, 2015), landscape-level forest type (Peters et al., 2022), surface geology (Soller et al., 2012), integrated soil moisture (Iverson et al., 1997), drainage (Schaetzl et al., 2009) and productivity (Schaetzl et al., 2012) indices, and elevation, slope aspect, and slope gradient from a 30 m digital elevation model (USGS, 2023). As in prior analyses, we aggregated NLCD classes into a smaller number of groups and inferred reforestation as pedons with forest cover and Ap horizons indicating past cultivation (Nave et al., 2019b, 2021, 2022a, 2022b, 2024). Also in keeping with prior analyses, we only extracted GIS attributes for pedons sampled 1989-present to ensure consistency between soil observations and GIS data.

As in prior published assessments, we used gap-filling to estimate Db for soil horizons in the NCSS Database that did not possess measured values. Using an initial dataset of $n = 4645$ individual horizons, we developed a relationship between measured Db and %C values for the fine earth fraction (soil passing a 2 mm sieve). For %C, we preferentially used the following: (1) %organic C measured directly, or (2) %total C minus %inorganic C, or (3) assumed %total C = %organic C, for samples with no measured %inorganic C but pH <7.0. We excluded soils with pH >7.0 and no reported value for %inorganic C as it would have been impossible to calculate %organic C in these cases. Next, we derived an exponential decay model (eq. [1]) that had $r^2 = 0.22$, $P < 0.001$, and a standard error of the estimate of 0.16, for soils with %C ranging from 0.00 to 16.2 % and Db ranging from 0.67 to 2.86 g cm^{-3}.

$$\mathrm{Db} = 1.0064 + 0.6867^{*}\, exp.(-0.2477^{*}\%\mathrm{C}) \tag{1}$$

For all mineral soil horizons in the pedon data, we then computed SOC stock (Mg C ha^{-1}) as the product of %C, Db, and horizon thickness. As in our prior papers, we focused our analyses on A horizons and whole soil profiles, summing individual horizons to obtain total C stocks (e.g., for A1 + A2 horizons, and for the many horizons comprising whole soil profiles). Because organic horizons were sporadically reported and had %C values well beyond the range of the input data for our Db - %C model, we excluded O horizons from our analyses of the pedon data and instead relied on NFI data.

2.5. National forest inventory dataset

We complemented our meta-analysis and NCSS pedon + GIS datasets with NFI data from the USDA FS. The NFI plots are the basis for what is widely known as the Forest Inventory and Analysis (FIA) program, which provides data from an equal-probability sample of all forest lands in the conterminous U.S. Across the conterminous U.S, there is one permanent plot on approximately every 2400 ha, randomly located within a hexagonal sampling frame (McRoberts et al., 2005). All NFI plots with at least one forest condition are measured every 5–7 years in the eastern U.S. 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 NFI spatial design ensures no systematic bias with regard to location, ownership, composition, soil, physiographic factors, or other characteristics. We obtained data for this analysis from a July 2022 query of the FIA Database for records of organic horizons, 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 the FIA Database (USDA FS, 2022) of each NFI plot, and only plots observed since 2000, to align with meta-analysis and NCSS pedon datasets. Similar to the NCSS pedon data, we used GIS lookups of NFI plot locations to obtain the following attributes, from the same sources: MAT, MAP, and climatic moisture index (CMI; PRISM, 2015), surface geology (Soller et al., 2012), and soil taxonomic order (Soil Survey Staff, 2022). Additional plot-level variables recorded by NFI included elevation, slope gradient and aspect, physiographic position, forest type, aboveground biomass, and disturbance and treatment histories. Altogether, our NFI datasets for the study area included $n = 416$ organic horizons and $n = 310$ mineral soils (Phase 3 or P3 plots).

2.6. Statistical analysis of NCSS and NFI data

To complement the meta-analysis of experimental data from published papers, we used inferential statistics (SigmaPlot 14, SYSTAT Software, San Jose, CA US) to analyze observational NCSS and NFI data. We identified factors influencing baseline C stocks in (1) O horizons, (2) A horizons, and (3) whole soil profiles using best subsets regressions to identify variables with statistically significant categorical or continuous relationships with SOC stocks. We analyzed O horizons from the NFI dataset because it was a larger source of information about this surface horizon, which is under-sampled in the NCSS dataset, and A horizons and whole profiles from the NCSS dataset, which provides better information about mineral soils (especially at depth). In these analyses, we coded categorical predictors as dummy variables and standardized continuous predictors by subtracting the mean and dividing by the standard deviation. We defined the optimal model as the one with the highest adjusted R^2 with at least a 0.5 % incremental increase in adjusted R^2 over the previous model, and comprised entirely of predictor variables with statistically significant *P* values. We set these criteria to balance between identifying the many factors that influence SOC stocks vs. protecting against over-fitting and developing overly complex models. The full suites of potential predictor variables assessed during model selection are reported in Table S1 (NFI O horizons) and Table S2 (NCSS A horizons and whole profiles). To test for land use differences in soil and site properties in the NCSS dataset, we did not attempt to normalize the strongly non-normal response data, and instead relied upon non-parametric analyses to identify significant differences (Kruskal-Wallis ANOVA on ranks with Dunn's Multiple Comparisons; Nave et al., 2018).

To evaluate meta-analytic trends using NFI data, we compared mean C stocks in NFI plots that were (vs. were not) harvested, and in plots that were (vs. were not) documented as having experienced fire, using *t*-tests. We limited these comparisons in two ways. First, we only analyzed data for O horizons, which were the only portion of the profile with significant underlying explainable variation according to meta-analysis. Second, for harvest and for fire, we only incorporated into the control group those NFI plots that had the same attributes as the NFI plots in the treatment group, for the following variables: forest type, soil order, physiographic class, slope aspect, and aboveground biomass (as a range). While the resulting statistical comparisons were still unbalanced for both forest harvest (control $n = 285$, treatment $n = 36$) and fire (control $n = 178$, treatment $n = 10$), we assumed that excluding control plots that differed in factors controlling baseline variation in C stocks (see Table 2) minimized the potential for a spatially confounded design bias in our inferences.

For all statistical analyses, we set $P < 0.05$ as the threshold for accepting results as significant.

3. Results

3.1. Baseline controls on soil C stocks

Best subsets regression model selection yielded two insights into baseline variation in soil C stocks across the study area. First, most of the existing variation in C stocks is unrelated to any of the (numerous) factors that we tested (Table 2). Second, although much of the variation in soil C stocks remains unexplained, a generally consistent set of factors emerged as significant predictors of the explained proportion.

In O horizons, only 16 % of the variance in C stocks was explainable according to the factors we included in model selection, and among these, forest type had the strongest influence. Compared to oak-hickory (the default forest type in the model), pine types (white and loblolly-shortleaf) had larger O horizon C stocks, and the elm-ash-cottonwood type had smaller O horizon C stocks. Soil taxonomy was also predictive, with Entisols (Regosols) having smaller C stocks than the default (Alfisols [Luvisols]), as was physiography: mesic uplands had smaller O horizon C stocks than the default physiographic group (mesic flatwoods), and south-facing slopes had larger C stocks than their corresponding default (north-facing slopes). Lastly, O horizon C stocks increased significantly with aboveground biomass C stocks.

For A horizons, the optimal model explained 22 % of the variance in C stocks, with only 3 variables. In that case, soil taxonomy was also important, with Mollisols (Phaeozems) having larger A horizon C stocks than the default order (Alfisols [Luvisols]). The remaining two predictors were continuous variables, with C stocks increasing with clay content and slope steepness.

Variation in soil C stocks was most explainable for whole soil profiles (23 %), according to many factors. Soil taxonomy was again important, with Mollisols (Phaeozems) having larger, and Ultisols (Acrisols) having

Table 2
Variables included in optimal best subsets regression models of O horizon ($R^2 = 0.16$), A horizon ($R^2 = 0.22$), and profile total ($R^2 = 0.23$) C stocks ($P = 0.001$ for each). Variables are sorted descending by their relative strength of influence (in terms of standardized slope coefficients). For each variable, the slope coefficient, standard error, *P* value, and variance inflation factor is reported. See Tables S1 and S2 for the complete list of variables included in the selection pool for this modeling procedure, and for correlation of USDA Soil Orders to World Reference Base Reference Soil Groups.

O horizon (NFI) $R^2 = 0.16$ $P < 0.001$					A horizon (NCSS) $R^2 = 0.22$ $P < 0.001$					Profile total (NCSS) $R^2 = 0.23$ $P < 0.001$				
Variable	Coef.	SE	P	VIF	Variable	Coef.	SE	P	VIF	Variable	Coef.	SE	P	VIF
Constant	1.98	0.04	<0.001	0.00	Constant	3.63	0.01	<0.001	0.00	Constant	4.68	0.01	<0.001	0.00
White-red-jack pine	0.87	0.30	<0.001	1.01	Mollisols	0.59	0.03	<0.001	1.17	Proglacial deposits	−0.51	0.04	<0.001	1.05
Elm-ash-cottonwood	−0.50	0.14	<0.001	1.04	Clay content	0.16	0.01	<0.001	1.16	Mollisols	0.45	0.02	<0.001	1.15
Loblolly-shortleaf pine	0.44	0.13	<0.001	1.02	Slope gradient	0.05	0.01	<0.001	1.01	Glacial till	−0.26	0.03	<0.001	1.05
Entisols	−0.42	0.16	0.01	1.02						Clayey-skeletal texture	0.19	0.03	<0.001	1.05
Mesic upland	−0.25	0.05	<0.001	1.07						Ultisols	−0.12	0.02	<0.001	1.44
South-facing slope	0.14	0.06	0.02	1.02						Fine-loamy texture	−0.12	0.02	<0.001	1.09
Aboveground biomass	0.14	0.03	<0.001	1.03						Profile depth	0.11	0.01	<0.001	1.11
										Noncalc. residuum	−0.09	0.01	<0.001	1.11
										Productivity index	0.07	0.01	<0.001	1.58
										Elevation	0.04	0.01	<0.001	1.26

smaller, C stocks than the default (Alfisols [Luvisols]). Soil texture was significant, with clayey-skeletal textures having larger, and fine-loamy textures having smaller C stocks than the default group (fine-silty soils). Additional soil property influences included positive relationships between profile total C stocks and (1) soil depth and (2) taxonomic productivity index. Physiographic variables were also significant, with profile total C stocks increasing with elevation, and varying according to parent material. Namely, proglacial deposits, glacial till, and non-calcareous sedimentary residuum all had smaller whole-profile C stocks than the default (calcareous sedimentary residuum).

3.2. Land use and management effects on soil C stocks meta-analytic main effects

Meta-analysis of published papers revealed significant effects of forest management, mine reclamation, and land use change on soil C stocks across the study area (Fig. 2). Forest management had small effects on soil C stocks, whether overall negative as in the case of prescribed fire, or overall positive as in the case of harvesting. The combination of harvest and prescribed fire had no significant effect on soil C stocks. Increases in soil C stocks occurred with all forms of mine reclamation, and were larger when reclaimed to forest or grassland than to agriculture. In terms of land use change, deforestation for mining and reforestation of agricultural lands were associated with decreased soil C stocks, while deforestation for agriculture was associated with increased soil C stocks.

3.3. Land use and management effects on soil C stocks: underlying meta-analytic trends

Meta-analysis revealed that there was substantial underlying variation in the soil C stock changes associated with each land use and management treatment. For forest management, we explored this variation for O horizons because they were well-replicated (k = 48) and much of this variance was explainable, whereas mineral soils had smaller between-group sample sizes and showed no explainable underlying variation according to any of the predictor variables examined. Among O horizons, prescribed fire and harvest+prescribed fire treatments decreased C stocks, while harvest alone had no influence (Fig. 3). Considering only O horizons from the two treatments that declined, (prescribed fire and harvest+prescribed fire treatments), response ratios calculated from horizon thickness or mass showed significant declines with treatment, while response ratios calculated from C concentration showed no change (Fig. 4). These trends indicate significance in the sense of individual groups of response ratios differing from 0 % change (see section 2.3), even though reporting units were not a statistically significant source of variation ($P = 0.09$). Several additional trends also emerged for O horizons from prescribed fire and harvest+prescribed fire treatments. First, studies that sampled the intact litter (Oi) or intact and partially decomposed litter (Oi and Oe) portions of the O horizon showed significant declines with prescribed fire, while studies that reported only the partially to well-decomposed portion of the O horizon (Oe and Oa; duff) showed no significant change. Spring burns were associated with O horizon declines, while winter and winter-spring burning regimes showed no significant change. Slightly drier (intermediate) physiographic settings showed O horizon declines with burning, while slightly moister (sub-mesic) settings showed no significant change. Lastly, prescribed fire was associated with O horizon declines in mixed mesic hardwood and oak-hickory forest types, but no change in pine-oak forests.

Studies of mine reclamation (k = 65) were entirely focused on mineral soils and had substantial variability ($Q_t = 118.4$), which varied according to several predictor variables. Parent material had the largest explanatory power ($Q_b = 31.6$), representing ca. 27 % of the total heterogeneity (Q_t). Of the three classes of parent material reported in the papers, coarse-loamy calcareous till showed the smallest soil C stock increase with post-mine reclamation (mean = +28 %), sandstone-derived overburden showed an intermediate soil C stock increase (+134 %), and mixed sandstone-shale overburden showed the largest increase (+548 %). The predictor variables with the second- and third-most explanatory power were mean annual temperature (meta-regression $Q_m = 25.6$, $Q_t = 118.4$) and ecological section ($Q_b = 22.7$, $Q_t = 117.4$), respectively. For temperature, the slope of the relationship was negative, indicating smaller soil C stock increases with reclamation in warmer climates, while for ecological section, increases were smallest for 223G (Central Till Plains – Oak-Hickory), intermediate for M221C (Northern Cumberland Mountains), and largest for 221E (Southern Unglaciated Allegheny Plateau).

Reforestation after agriculture, which was associated with an overall meta-analytic soil C stock decline (k = 76, all mineral soils), had no

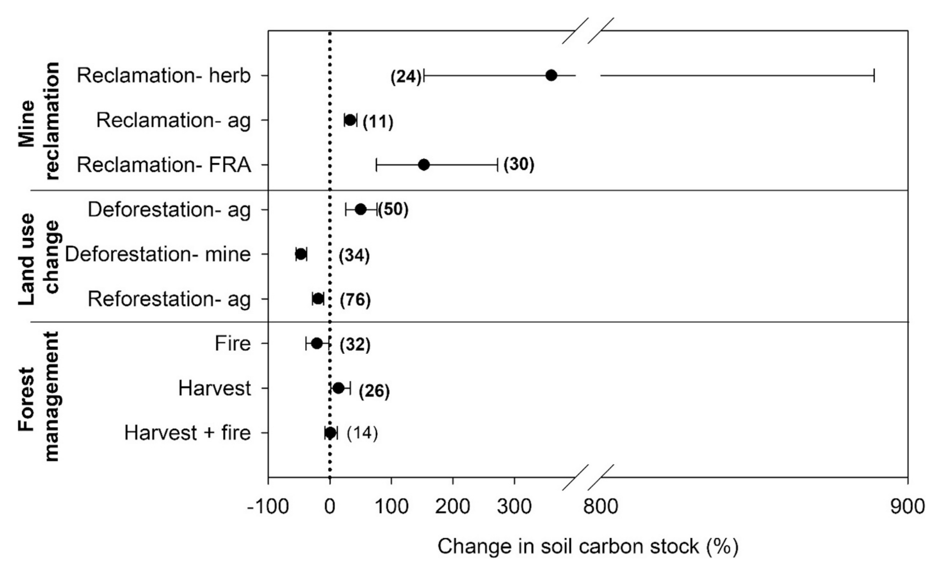

Fig. 2. Main effects of land use and management on soil C stocks (all response ratios), according to meta-analysis. The plot shows the mean effect size, bootstrapped confidence interval, and sample size for each treatment. The dotted vertical reference line indicates 0 % change; treatments that do not overlap this reference line are indicated with bold sample sizes. Note: for mine reclamation, categories are reclamation with herbaceous vegetation (herb), agriculture (ag), or the forestry reclamation approach (FRA).

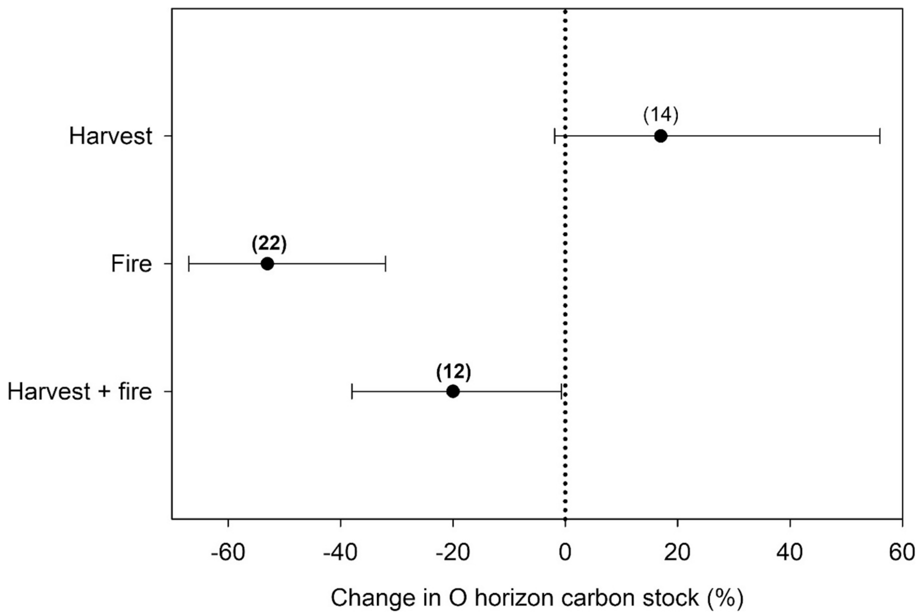

Fig. 3. Forest management effects on organic horizon C stocks, according to meta-analysis. The plot shows the mean effect size, bootstrapped confidence interval, and sample size for each treatment. The dotted vertical reference line indicates 0 % change; rows that did not overlap this reference line are indicated with bold sample sizes.

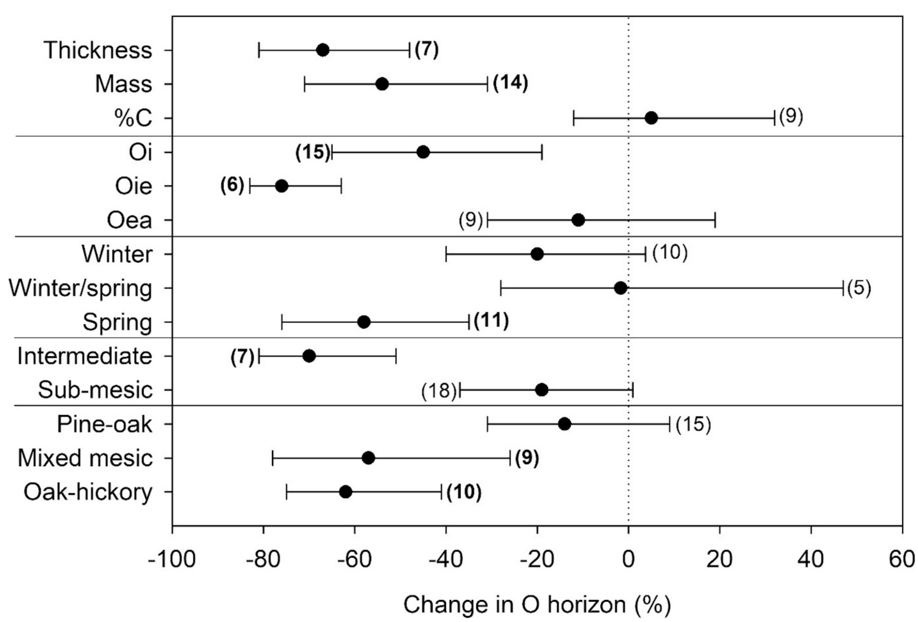

Fig. 4. Effects of forest management treatments involving prescribed fire (prescribed fire and prescribed fire + harvest) on organic horizon C stocks, as a function of underlying predictor variables, according to meta-analysis. The plot shows the mean effect size, bootstrapped confidence interval, and sample size for each treatment group. The dotted vertical reference line indicates 0 % change; treatment groups that do not overlap this reference line are indicated with bold sample sizes. See section 3.3 for grouping variable definitions.

statistically significant predictor variables, but did show several individual groups of response ratios differing from 0 % change (Fig. 5, top). Planted forest stands were associated with significant soil C stock reductions, while naturally invaded old fields showed no soil C effects of reforestation. Ecoregions also differed in their responses, with studies from 221E and 223D (Southern Unglaciated Allegheny Plateau and Interior Low Plateau – Shawnee Hills, respectively) showing SOC losses, and 223F (Interior Low Plateau – Bluegrass) showing no significant difference from 0 % change. Soil C stocks in the soil surface (0–10 cm) were not significantly different from 0 % change, while observations from deeper samples (10–30 cm, >30 cm) showed soil C stock declines.

Deforestation due to a land use change to agriculture was associated with an overall meta-analytic soil C stock increase ($k = 50$, all mineral soils) and varied with several statistically significant predictor variables (Fig. 5, bottom). First, individual publications differed in their reported soil C stock changes; of those with $k \geq 5$ response ratios, results from Walia et al. (2017) indicated soil C gains with deforestation, while results from Nath and Lal (2017) indicated soil C declines. Ecoregions also differed, with 221E (Southern Unglaciated Allegheny Plateau) showing soil C declines with deforestation, and 223D (Interior Low Plateau – Bluegrass) showing a soil C increase. The effects of soil sampling depth were the reverse of reforestation after agriculture, with surficial (0–10

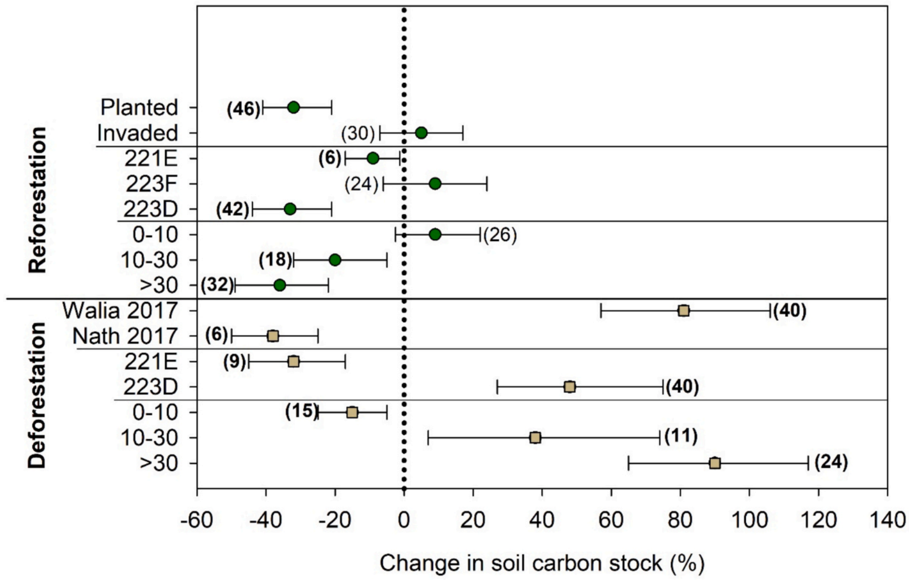

Fig. 5. Effects of land use change on mineral soil C stocks, as a function of underlying predictor variables, according to meta-analysis. Green points in the top half of the figure are for reforestation after agriculture; brown points in the bottom half are for deforestation to agriculture. The plot shows the mean effect size, bootstrapped confidence interval, and sample size for each treatment group. The dotted vertical reference line indicates 0 % change; treatment groups that do not overlap this reference line are indicated with bold sample sizes. See section 3.3 for grouping variable definitions.

cm) soil C stocks declining, and deeper (10–30 and > 30 cm) soil C stocks increasing with deforestation.

3.4. *Land use and management effects on soil C stocks: evaluation with observational data*

Data from the NCSS and NFI programs afforded several opportunities to evaluate trends revealed by meta-analysis. The NCSS pedon + remote sensing data allowed a comparison of (1) pedons under active agriculture, (2) pedons that were cultivated in the past but have been subsequently reforested, and (3) pedons from never-cultivated forest lands (Fig. 6). Across these three land uses, nearly every soil, topographic, and ecosystem property assessed showed significant differences, with agricultural soils and never-cultivated forest soils representing the extremes, and reforesting soils having intermediate values. In lands under active agriculture, A horizons were thickest and had the largest C stocks, clay contents, productivity and drainage indices, the highest pH, and the gentlest slopes and least aboveground biomass of the three land uses. Forest pedons had the thinnest A horizons, were the most strongly acid and sandy, occurred on the steepest slopes, had the lowest productivity and drainage indices, and had the most aboveground biomass. Forest pedons had significantly smaller A horizon C stocks than agricultural soils, but did not differ from reforesting soils. Reforesting pedons were intermediate between (and significantly different from) permanent forest and agricultural pedons in all other properties. Comparisons of NFI plots that were vs. were not recently harvested, and were vs. were not recently burned, showed no statistically significant differences in O horizon C stocks ($P > 0.05$).

4. Discussion

4.1. *Baseline variation in soil C stocks*

This analysis, focused on the Central Hardwoods region of the U.S., builds on our prior assessments for other ecoregions in two ways. First, we show that a different set of factors regulates baseline variation in soil C stocks in this study area than in other ecoregions. Second, we show how the land uses and management treatments typical of the Central Hardwoods influence its soil C stocks in ways that cannot be directly transferred from patterns in other ecoregions, or generalized from broader-scale literature reviews. This underscores that while there is global interest in climate change mitigation through soil C management, there is no single globally appropriate approach.

Drivers of spatial variation in soil C stocks highlight the Central Hardwoods as a region of many ecological gradients. For example, two pine forest types with opposite continental distributions—one northern and dominated by white pine, the other southern and dominated by loblolly pine—are both associated with large O horizon C stocks. This relationship between forest type and O horizon C stocks, as well as the positive relationship between O horizon C stocks and aboveground biomass, collectively imply a biotic control of O horizon C stocks, potentially via the quantity and quality of litter inputs (Washburn and Arthur, 2003; Polyakova and Billor, 2007). At the other extreme from the small, superficial C stocks of the O horizon, variation in C stocks in whole soil profiles is largely influenced by bottom-up factors. Owing to its diverse geology, soils of the Central Hardwoods have formed in parent materials ranging from glacial and eolian sediments, to alluvial, colluvial, and residual materials derived from very different bedrock types (Lee and Kabrick, 2017). These parent materials influence whole-profile C stocks, and are likely the basis for relationships between C stocks and other soil properties, such as texture, taxonomic order, and taxonomy-based productivity index.

At nearly 600,000 sq. km, the Central Hardwoods is among the largest of the six ecoregional assessments we have prepared to date. However, despite its large extent, it spans narrower climate gradients than our other assessments, and unlike all other ecoregions, no climate variables emerged as statistically significant predictors of variation in soil C stocks. In all other ecoregions, soil C stocks have varied with temperature, precipitation, or both. This uniqueness extends to the overall predictability of soil C stocks across the six study regions. Whereas multiple regression models in our other ecoregional assessments have explained 32–51 % of the variation in surface soil and whole profile C stocks, r^2 values for the Central Hardwoods indicate predictive relationships that are roughly half as powerful (Table 2). This lack of a climatic influence and lower predictability of soil C stocks may be a statistical artifact of the narrower range in temperature and precipitation in the Central Hardwoods, or may reflect greater underlying variation in other factors, such as parent material. Regardless, it is clear that

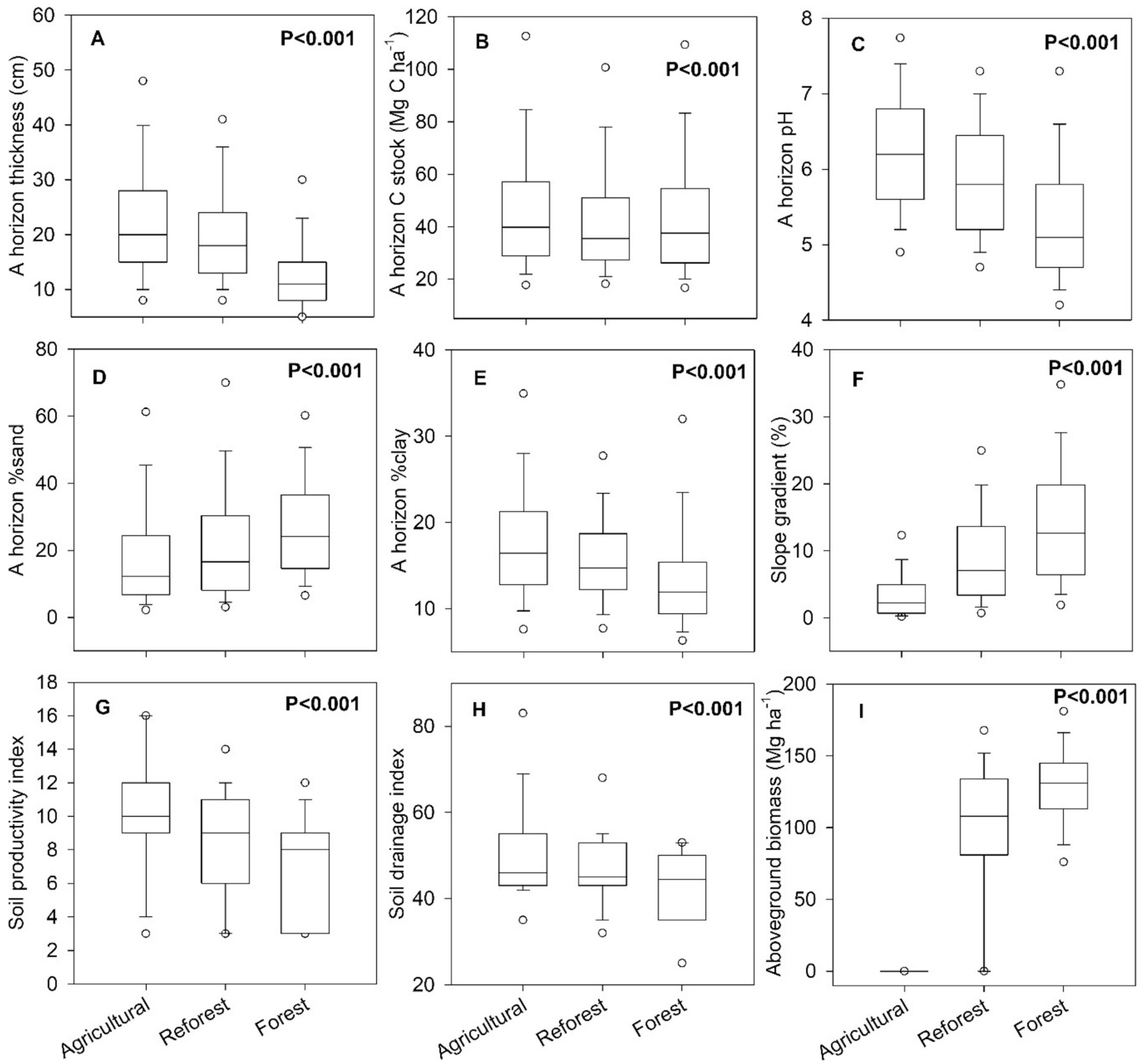

Fig. 6. Soil and site properties as a function of contemporary land use. Boxplots show medians and the 5th, 25th, 75th, and 95th percentiles for each of three land uses: (1) active agricultural lands [Agriculture], (2) previously cultivated agricultural lands now under forest cover [Reforest], and (3) never-cultivated forest lands [Forest]. *P* values are for Kruskal-Wallis ANOVA on ranks. Differences between medians are statistically significant between all three land uses for every variable except SOC stock, in which only agriculture and forest differ from each other.

generalization from global or other regional literature, such as climatic control of soil C stocks, would be inappropriate in the Central Hardwoods ecoregion as we define it here.

4.2. Forest management effects on soil C stocks

Moving from baseline variation to management effects, the Central Hardwoods is among the best-studied regions we have assessed to date in terms of fire effects on soil C stocks. This reflects well-established prescribed fire and ecosystem restoration programs in the oak-hickory and oak-pine forests of the region. Most of this literature focuses on tree regeneration, biodiversity, wildlife, and other management goals (Hutchinson et al., 2005; Iverson et al., 2017; Kabrick et al., 2008; Harper et al., 2016; Schweitzer et al., 2016; Blankenship et al., 2023). However, in our literature review and meta-analysis, a sufficient number of papers reported fire effects on O horizons to generate several ecologically consistent insights (Fig. 4). First, reductions in O horizon thickness and mass, without changes in C concentration, are evidence that prescribed fires diminish O horizons by combusting organic matter, not by changing its chemical characteristics (as %C). This is a reminder that when accounting for fire effects on soil C stocks, emphasis should be on the direct losses of C-containing organic matter, rather than on potential changes in the chemical composition of the residual material (Pellegrini et al., 2021). Significant declines in Oi and upper Oe subhorizons, but not in lower Oe or Oa subhorizons, indicate that prescribed burns in the region are generally effective at reducing the less decomposed surface litter component of the O horizon, while leaving the deeper, more decomposed (and typically moister) duff intact. This is in keeping with typical burn prescription goals, which emphasize retaining the more decomposed portions of the O horizon to protect the mineral soil surface and for soil moisture and nutrient supply, while reducing surface litter, which inhibits tree seed germination and can fuel more severe wildfires (Loucks et al., 2008; Arthur et al., 2017) This tendency for Oi / Oe horizon losses may also be reflected in the influence of forest type. Oak-hickory and other hardwood forest types generally have less developed O horizons than forests with pine (Table 2), and relatively more of their O horizon consists of higher-quality, faster-turnover organic matter: Oi and Oe material (Li et al., 2009). The smaller mass and more intact condition of hardwood O horizon litter would make them more likely to lose a larger share of their O horizon mass, while the deeper O horizons of pine, including a substantial Oa subhorizon (duff layer), would experience a relatively smaller reduction in overall O horizon mass from a fire of the same intensity. It is also important to recognize that the Oi and Oe horizons can recover quickly during fire-free years. In the Ozark Highlands Central Hardwood Region, Stambaugh et al. (2006) estimated that 75 % of the litter is recovered within four years after burning and 99 % is recovered after a decade. This rapid recovery is important in light of the available literature on this topic: of the 8 fire studies available for meta-analysis, 6 reported results from <5 years since prescribed fire.

Fire effects on soil C stocks in the Central Hardwoods are largely different from fire effects on soils in other ecoregions. In all regions with enough fire studies to support our ecoregional meta-analyses, O horizons have shown significant declines, with mean C stock reductions of 45–70 % (Nave et al., 2019b, 2021, 2022a, 2022b). Organic horizon reductions in the Central Appalachians fall within this range. There, however, the similarities largely end. Whereas mineral soil C stocks in the Central Hardwoods are not affected by fire, mineral soils in other ecoregions showed a range of responses. In the Lake States, fires decreased C stocks in A and E horizons, while increasing C stocks in deeper horizons (Nave et al., 2021). In the Pacific Northwest, fires decreased A and B horizons (Nave et al., 2022a). In the South Atlantic states, fires had no effect on surface mineral soils, but increased C stocks deeper in the mineral soil (Nave et al., 2022b). Only in the Central Appalachians / Mid-Atlantic ecoregion did soil C stocks show identical trends as in the Central Hardwoods, with O horizon declines and no net change in mineral soils (Nave et al., 2019b). The fact that O horizons show consistent declines across regions, generally increasing with fire severity, suggests that this finding is broadly applicable across forest soils. However, management effects on mineral soil carbon pools are much more ecoregion-specific, potentially due to factors such as soil parent material, texture, climate, and forest type. This regional variability in mineral soil responses indicates it is inappropriate to expect the same type of disturbance or management to have the same outcomes everywhere. Regional variability in soil C outcomes also highlights the wide variation in fire severity among different kinds of ecosystems across the globe (Certini, 2005).

4.3. Mine reclamation and soil C stocks

Across the Central Hardwoods and into the Appalachian Mountains of the eastern U.S., coal mining has affected thousands of square kilometers in the past 1–2 centuries (Amichev et al., 2008; Avera et al., 2015). In the last 50 years, reclamation of these lands has frequently utilized the Forestry Reclamation Approach (FRA), which involves planting trees on the landforms and soils reconstructed from overburden after coal removal (Zipper et al., 2011). The addition of trees may influence total ecosystem C recovery compared to herbaceous vegetation alone (e.g., grass-legume mixtures), but our meta-analysis suggests that these two restoration approaches have similar soil C outcomes. In contrast, parent material does appear to have a significant effect on post-reclamation soil C stocks, with shale-sandstone mixtures showing larger soil C stock gains than pure sandstone spoils. Mine reclamation literature from the eastern U.S. suggests this may be because spoils containing shale are less acidic and have greater clay and short range-order mineral contents, which support greater water holding capacity and vegetation productivity than pure sandstone spoils (Miller et al., 2012; Sena et al., 2018). Furthermore, shale-derived spoils can contain petrogenic organic matter that increases vegetation productivity through similar mechanisms (Copard et al., 2007; Rumpel et al., 1998). Broader research on undisturbed ecosystems also supports this pattern, with shale soils from around the world showing larger C stocks than co-occurring sandstone soils (Aguilar and Heil, 1988; Choudhury, 2023), and ecosystems on shale maintaining higher rates of primary production than adjacent systems on sandstone (Reed and Kaye, 2020). Regardless of the mechanisms involved, our results show that where mining has occurred but reclamation has not, C outcomes may be improved if pure sandstone spoils are amended with shale materials. In areas where shale spoils are not available, ameliorative steps or additional reclamation practices such as liming or surface mulch application may produce longer-term soil C benefits through positive feedbacks with vegetation productivity. Where reclamation has already occurred, our results may help inform management of the recovering ecosystems, based on the influence of their underlying bedrock geology on their soil C stocks.

4.4. Land use change and soil C stocks

Our results demonstrate how soil properties have affected land use across the region in the past, and continue to do so today. Harmonized NCSS + remote sensing data reveal that the most productive soils with the most favorable physical properties are preferentially used for cultivation, the least productive and most physically unfavorable soils have remained under forest cover, and intermediate soils are where reforestation tends to occur (Fig. 6). As expected from these differences in physical properties, agricultural soils hold more C than permanent forest soils, while C stocks in reforesting soils are not significantly different from either agricultural or permanent forest soils. In the context of these pre-existing differences in soil physical properties, meta-analysis results for soil C stocks are ambiguous. On one hand, the meta-analytic increase in soil C stocks with deforestation could be an artifact of the experimental designs of the published studies. On the other hand, several potential mechanisms reported in the literature could explain soil C gains with deforestation or losses with reforestation. First, after deforestation, tillage mixes C-rich surface soils to depth, depleting C at the surface while increasing it at depth. This pattern is revealed both in our analysis (Fig. 5) and in other meta-analyses (e.g., Priori et al., 2024). Second, long-term organic matter and nutrient additions to agricultural soils through manure or fertilizer can increase C stocks above and beyond baseline forest levels (Tian et al., 2015; Jagadamma et al., 2007). Third, under reforestation, aggrading forests on formerly cultivated soils can actively degrade or "mine" deep soil organic matter in order to liberate mineral nutrients, depleting C stocks at depth while supporting rapid aboveground C accumulation in vegetation (Yang et al., 2022; Yang and Knops, 2023). Losses may be especially acute for planted versus naturally established forests, as implied by results of our meta-analysis (Fig. 5) and others (Liao et al., 2012). In the context of this broader literature, our results imply that soil C stocks under agricultural, forest, and reforesting land uses reflect both pre-existing differences between land uses and changes in processes that accompany land use change. This precludes quantifying rates of soil C change with confidence, but may facilitate future comparisons of soil C stocks across land uses, with experimental designs that control for soil physical properties.

4.5. Conclusions

Soil C stocks across the Central Hardwoods are controlled by biotic and abiotic factors, most notably forest type and soil parent material. Among contemporary management tools, forest managers' ability to change soil C stocks is essentially limited to prescribed fire, which diminishes O horizons, to an extent determined by place (ecological factors such as forest type) and practice (burn prescription details such as timing). Reclamation of barren post-mining substrates leads to large soil C gains (compared to a starting point that has little to no C), though the presence or absence of trees appears to have no effect on soil C recovery. Rather, variation in substrate influences soil C stock increases after mining, with sandstone+shale mixtures being associated with larger soil C increases than sandstone alone. Where soil C recovery is a management goal, reclamation decisions may thus be more effectively guided by geologic than biotic considerations. Effects of past and current land use and land use change on soil C stocks cannot currently be disentangled.

Our results suggest that agricultural soils hold more C than forest soils because they have more favorable properties to begin with, not because they are under agricultural use. Further studies of land use change and agricultural soil management are needed to constrain their C cycle and climate mitigation effects. Overall, land use change and forest management effects on soil C stocks in the Central Hardwoods cannot be extrapolated from individual site-level studies and are largely distinct from trends in other ecoregions. This highlights the need for a regionalized approach to such assessments to make global progress towards climate change mitigation through soil C management.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

This work was supported by the USDA Forest Service, Northern Research Station (agreement 22-CR-11242306-022). The authors thank Brian Walters for FIA data processing, Stephanie Connolly and Kate Heckman for insight into mining practices and soil properties in the region, and Maria Janowiak and Darren Miller for helpful comments on the manuscript draft. The authors are also grateful to the individuals who participated in the peer review process for identifying areas for improvement.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.geodrs.2025.e00930.

Data availability

Data will be made available on request.

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