New Forests (2019) 50:115–137
https://doi.org/10.1007/s11056-018-9655-3

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The role of reforestation in carbon sequestration

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Received: 12 October 2017 / Accepted: 24 June 2018 / Published online: 9 July 2018

Abstract
In the United States (U.S.), the maintenance of forest cover is a legal mandate for federally managed forest lands. More broadly, reforestation following harvesting, recent or historic disturbances can enhance numerous carbon (C)-based ecosystem services and functions. These include production of woody biomass for forest products, and mitigation of atmospheric CO_2 pollution and climate change by sequestering C into ecosystem pools where it can be stored for long timescales. Nonetheless, a range of assessments and analyses indicate that reforestation in the U.S. lags behind its potential, with the continuation of ecosystem services and functions at risk if reforestation is not increased. In this context, there is need for multiple independent analyses that quantify the role of reforestation in C sequestration, from ecosystems up to regional and national levels. Here, we describe the methods and report the findings of a large-scale data synthesis aimed at four objectives: (1) estimate C storage in major ecosystem pools in forest and other land cover types; (2) quantify sources of variation in ecosystem C pools; (3) compare the impacts of reforestation and afforestation on C pools; (4) assess whether these results hold or diverge across ecoregions. The results of our synthesis support four overarching inferences regarding reforestation and other land use impacts on C sequestration. First, in the bigger picture, soils are the dominant C pool in all ecosystems and land cover types in the U.S., and soil C pool sizes vary less by land cover than by other factors, such as spatial variation or soil wetness. Second, where historically cultivated lands are being reforested, topsoils are sequestering significant amounts of C, with the majority of reforested lands yet to reach their capacity relative to the potential indicated by natural forest soils. Third, the establishment of woody vegetation delivers immediate to multi-decadal C sequestration benefits in aboveground woody biomass and coarse woody debris pools, with two- to three-fold C sequestration benefits in biomass during the first several decades following planting. Fourth, opportunities to enhance C sequestration through reforestation vary among the ecoregions, according to current levels of planting, typical forest growth rates, and past land uses (especially cultivation). Altogether, our results suggest that an immediate, but phased and spatially targeted approach to reforestation can enhance C sequestration in forest biomass and soils in the U.S. for decades to centuries to come.

Keywords Forest ecosystem · Land cover · Land use · Soil · Biomass · ECOMAP

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Introduction

In the United States, there is a legal mandate to maintain forest cover on designated forest lands managed by the U.S. Department of Agriculture, Forest Service (USDA-FS). The earliest version of this mandate dates to the Forest Service Organic Administration Act of 1897, and has been reinforced repeatedly by Acts of Congress many times since the initiation of the Forest Service (1911, 1930, 1949, 1974, 1976, and 1980). At the agency level, numerous internal directives in the Forest Service Manual (https://www.fs.fed.us/im/directives/) specify guidance for management activities intended to maintain, regenerate, or restore forest cover, and reforestation is one of the most important of these activities on the 77 million hectares comprising the National Forest System (NFS). However, land area targets for reforestation on NFS lands have been under-attained by 75–85% for at least 15 years, partly due to insufficient infrastructure and funding for forest management activities, resulting in a widening gap between required and realized reforestation goals (Watrud et al. 2012). Unless reforestation is increased at a national level, concerns such as the uncertain longevity of the U.S. forest sector carbon (C) sink (Birdsey et al. 2006; Zhang et al. 2012; Oswalt et al. 2014; USDA Forest Service 2016), forest area decline (Yang and Mountrakis 2017), increases in forest disturbance extent and severity (Bentz et al. 2010; Kurz et al. 2008; Schoennagel et al. 2017), or interactions with ongoing climate change (Hicke et al. 2012; Liang et al. 2017) will only magnify.

The role of forests in mitigating atmospheric CO_2 pollution and climate change provides long-term context, and argues for a closer look at intensified reforestation efforts in the U.S. (Dumroese et al. 2015). In terms of context, the U.S. forest sector is providing a tremendous, but slowly diminishing ecosystem service by acting as a long-term net C sink, driven largely by forest regrowth following widespread historic disturbances (Caspersen et al. 2000; Williams et al. 2012, 2014). In terms of justification, not only is reforestation necessary to promptly re-establish forest cover after catastrophic disturbances such as large wildfires, but deliberate reforestation—even after less severe or extensive disturbances—may enhance C sequestration rates compared to passive management approaches such as waiting for natural regeneration (MacDonald et al. 2015; Nave et al. 2018; Post and Kwon 2000; Sample 2017). Across many regions and types of stand-replacing disturbances, even re-growing forests are net C sources to the atmosphere for a period of years to decades (Bond-Lamberty et al. 2004; Gough et al. 2007; Kashian et al. 2006; Law et al. 2003). Shortening the duration of this period during which ecosystem C outputs (e.g., through heterotrophic respiration) exceed ecosystem C inputs (e.g., through primary production) equates to a more positive C balance (i.e., greater storage) over the lifetime of the stand, and one obvious way to do so is accelerate canopy closure by ensuring adequate stocking density in the re-growing stand.

The scale and scope of the problems facing U.S. forests—in particular, the increasing area of disturbances and chronic reforestation shortfall—call for multiple evaluations, projections, and predictions of the C cycle implications of reforestation (or its neglect). In the present study, offered as complementary to the many recent, regional to national-scale reviews and projections of the forest sector C balance (e.g., Coulston et al. 2015; Creutzburg et al. 2017; Jin et al. 2017; Oswalt et al. 2014; Puhlick et al. 2017; Wear and Coulston 2015; Woodall et al. 2015), we use empirical data, statistical analyses, and ecoregional scaling to quantify the impacts of reforestation on C sequestration at broad levels. We address this overall goal via four specific objectives in this study, which uses space-for-time substitution to compare C stocks on lands differing in their use: (1) estimate C stocks of major ecosystem pools in forest and other land cover types, in order to contextualize forests as C sinks; (2) quantify sources of variation in ecosystem C pools, focusing on regional patterns and drivers; (3) compare C pools among lands differing in past and present land use, thereby inferring impacts of forest loss, reforestation, and afforestation; (4) for all objectives, assess how results scale across ecological (rather than political) units.

Methods

Approach

We approached this work using several large data sets and sources, described in detail in the following subsections and in Nave et al. (2018). The first source, the 3rd generation version of the International Soil Carbon Network (ISCN) Database (Nave et al. 2017), is a database containing geographic, physical, chemical, and ecological data for > 433,000 individual soil layers (horizons or sampled depth increments) worldwide. Individual soil layers are the constituents of soil profiles; profiles are from one to many (> 10) per site, and most sites are georeferenced. Data in the ISCN Database were derived from 39 datasets contributed by individual investigators, research networks, and U.S. government agencies. The second principal data source in this analysis consists of "overlay data;" these are point-specific attributes, extracted from remote sensing data products, for the geographical coordinates of individual ISCN sites. Overlay data utilized in this analysis include: (1) land cover attributes from all four versions (1992, 2001, 2006, 2011) of the National Land Cover Dataset (Vogelmann et al. 2001; Homer et al. 2004, 2015; Fry et al. 2011), a LANDSAT-derived, 30 m resolution data product; (2) estimates of aboveground biomass C stocks from the National Biomass Carbon Dataset for the year 2000 (NBCD2000; Kellndorfer et al. 2013), also a 30 m data product. The third major data source for our analysis was the USDA-Forest Service, Forest Inventory and Analysis Database (FIADB; https://apps.fs.usda.gov/fia/datamart/). FIADB is the central source for systematically collected as part of the National Forest Inventory (NFI) program (http://www.fia.fs.fed.us); for the present analysis we report aboveground biomass and coarse woody debris data derived from the NFI plot inventory network.

ISCN data handling and C stock estimates

We began our work with the ISCN DB using the map-based data retrieval tool on the ISCN website <soilcarb.net> to download essential geolocation, descriptive, physical, chemical, and data contributor information for 319,316 individual soil layers from 52,178 unique profiles contained in a polygon completely surrounding the conterminous U.S. (CONUS). Importantly, given the variety of motivations and sampling designs represented by the contributors of these data, the dataset we downloaded as a starting point does not constitute a random nor systematic sample of soils in the U.S.; on the other hand, however, given its large size and origins from many data contributors, there is no a priori reason to assume that it is not representative of the range of soils in the U.S. Data used in this analysis are from sources including the USDA-Natural Resources Conservation Service (specifically, the September 2014 version of the National Soil Survey Laboratory's Soil Characterization Database); the U.S. Geological (Survey Site-Specific Soil Carbon Database for Mineral Soils of the Mississippi River Basin; Buell and Markewich 2004), the USDA-FS (Database for Landscape-scale Carbon Monitoring Sites; Cole et al. 2013), and several projects by individual researchers (Heckman et al. 2009, 2013; Nave and Nadelhoffer unpublished). Beginning with these 52,178 profiles, we proceeded through a series of filtering steps to eliminate those that were of non-CONUS or unknown geographic location, sampled prior to 1989, or had layers with bulk density values in excess of 2.65 g cm^{-3} or C concentrations greater than 60% (mass/mass). Our intent with these filters was to include only soils that were of known CONUS origin, were sampled reasonably concurrently with the remote sensing overlay datasets (see "Land cover and biomass overlay data"), and had individual layers with realistic bulk density values and C concentrations. We also harmonized layer and profile depths to a common standard in which the top of the profile (0 cm depth) was equal to the top of the O-horizon, as some contributed profiles were sampled with a zero reference equal to the top of the mineral soil and O-horizon depths entered as negative values. Additionally, we created a decision tree and used gap-filling to compute C concentrations for the maximum number of layers possible. Specifically, we used the organic C concentration as the preferred metric of C concentration; for layers missing this parameter, we derived a prediction equation based on total C concentration and inorganic C concentration to predict the organic C concentration (all in per cent by mass). For those layers missing inorganic C concentration, we assumed that the organic C concentration was equal to the total C concentration. Layers having no C concentration data were not usable for C stock calculations. Similarly, we developed a decision tree and used gap-filling to estimate missing bulk density values. We used the fine earth bulk density (mass of soil materials < 2 mm per volume of soil materials < 2 mm) as the preferred metric for layers possessing multiple variant forms of bulk density; for layers missing this parameter we used the whole soil bulk density (mass of all soil materials per volume of all soil materials) if available, and otherwise used predictions generated by USDA-NRCS (Sequeira et al. 2014) as estimates if no measurements were available. Overall, 50% of the soil layers possessed measured bulk density values and 50% were gap-filled using the published prediction equations. After computing the C stock of each soil layer as the product of its C concentration (%), bulk density (g cm^{-3}), and thickness (cm), and scaling to Mg C ha^{-1}, we summed the individual layer C stock values up to the whole profile level (the maximum sampled depth as reported by the data contributor). Throughout this process, we repeatedly checked our calculations, compared our assembled datasets against the originally downloaded source data and against previous, internally versioned files, in order to ensure consistency, repeatability, and quality of the data used in subsequent analyses. After completing all steps, we were left with 22,847 profiles meeting the criteria specified above.

Land cover and biomass overlay data

Our intent with remote sensing overlay data was to derive land cover and biomass information for the ISCN profiles (and their individual layers) described in "Land cover and biomass overlay data" section. For this reason, our first step in deriving overlay data was to exclude profiles sampled before 1 January 1989 (as described in "Land cover and biomass overlay data" section), and our second was to assign each profile to its closest (in time) NLCD product. Specifically, we assumed that the land cover type for soil profiles sampled between 1 January 1989 and 31 December 1996 was reasonably represented by the NLCD 1992 product; soil profiles from 1 January 1997 to 31 December 2001 were represented by NLCD 2001; soil profiles from 1 January 2002 to 31 December 2006 by NLCD 2006; soil profiles from 1 January 2007 to present (2014) by NLCD 2011. Thus, all soil profile sampling dates were within 3–4 years of their derived land cover dates. Previously, we successfully employed this conceptual approach on the ISCN 2nd generation DB in an assessment of afforestation effects on soil C in top soils of the U.S. northern prairie states, but did not explicitly test concurrence of the remotely sensed versus directly observed land cover information (Nave et al. 2013). In the present analysis, we chose to combine soil profiles in generally similar land cover types recognized as distinct by NLCD into major land cover groups (e.g., Mishra and Riley 2015), in order to increase within-group sample sizes and decrease the number and complexity of multiple comparisons in statistical analyses. Specifically, we placed all developed lands (high intensity, medium intensity, low intensity, and open space) into a single category (developed lands); pasture/hay and grassland cover types into a pasture/grassland group; different forest types (evergreen, deciduous, mixed) into a single forest group; wetland land cover types (herbaceous, woody, and water) into a single wetland category. Next, before proceeding with further data manipulations or analyses, we validated a subset of the NLCD classifications using observed profile vegetation notes (as provided by ISCN data contributors) for the 674 profiles possessing this information. Based on general familiarity with the various plant common and scientific names, taxonomic codes, and ecosystem classifications used by data contributors, we were able to interpret the vegetation notes for 71% (479) of these profiles. Of these, 79% (379) had vegetation observations consistent with the NLCD groups specified above and 9% were obviously incorrect, reflecting a spatial or temporal mismatch between the ISCN profile and the NLCD data. The remaining 12% misclassified low density or low stature forest vegetation types as shrub/scrub or vice versa. For this reason, we combined forest and shrub/scrub land cover types into a single land cover group (woody vegetation) for several of our statistical analyses.

From the NBCD2000, we extracted aboveground woody biomass densities (AGWB; Mg C ha^{-1}) for ISCN profiles associated with NLCD 2001 or 2006 land cover data, in order to ensure that the biomass values (which are themselves derived values based on remote sensing, NFI training plot data, and algorithms) were closely concurrent with the date of soil profile sampling. For both land cover and biomass datasets, we used ArcGIS (ESRI, Redlands, CA USA) to assign NLCD codes and biomass C stocks to each ISCN location.

Aboveground C stocks from FIADB

The NFI plots that are the basis for FIA data derive from an equal-probability sample of forestlands across the CONUS. There is one permanent plot on approximately every 2400 ha across the U.S., with each plot placed randomly within a systematic hexagonal grid (Bechtold and Patterson 2005). Sampling of each plot is conducted on fixed area subplots that vary in size depending on the metric, with inventory of canopy-level trees (> 12.7 cm dbh) being conducted on four 0.016 ha subplots. This design across the CONUS ensures that NFI data have no systematic bias with regard to forestland location, ownership, composition, soil, physiographic or other factors. For this analysis, we queried the FIADB for records of the mass density (Mg C ha^{-1}) of AGWB (derived from individual tree measurements and allometric equations) and coarse woody debris (CWD; derived from quadrat measurements of CWD piece volume and decay class, and estimates of CWD density). We acquired these C pool sizes for all single-condition plots in CONUS, i.e., only plots that are not divided along sharp boundaries into conditions of different stand age, slope, wetness, etc. These sources of localized (within-plot) variability complicate plot data interpretation and may introduce edge effects; furthermore, given the enormous number of NFI plots available we felt this decision was a reasonable way to exercise stringent control on data quality in our analysis. As an additional constraint, we only utilized the most recent observation of each long-term NFI plot, and only plots observed since 2000, in order to make NFI data reasonably concurrent with the ISCN soil C and overlay land cover and biomass data described above. In contrast to ISCN data, we did not gain access to nor require precise geolocations of NFI plots (which are legally obscured). Our analyses test variation in AGWB and CWD C stocks against predictors including forest age, establishment type, and ecoregion; because these are internally recorded attributes associated with each NFI plot in the FIADB, there is no particular need for highly localized geographic coordinates. Altogether, our datasets for AGWB and CWD consisted of 81,673 and 22,043 plots, respectively.

Ecoregional framework

All of the recent, insightful large-scale assessments of forest C storage in the U.S. have reported regional variation according to politically defined spatial units, such as individual states or arbitrary multi-state regions. While the subdivision of space along political boundaries can have a legitimate basis, such as a legal directive for a specific assessment, we chose in the present analysis to utilize an ecoregional framework to explore spatial variation in land cover and use, ecosystem and forest C stocks. In particular, we used ECOMAP, an effort initiated by the USDA-Forest Service in the 1990s to organize the U.S. land base into a hierarchical structure of ecological units (Cleland et al. 1997; McNab et al. 2007). ECOMAP is a framework, subject to ongoing refinement, that is intended to identify ecologically scalable spatial units for planning and management purposes. Because there are fundamental climatic, geologic, and other natural constraints that affect forest growth and C storage heedless of political boundaries, an ecological basis for scaling may be quite useful to silviculturists, nursery managers, and others interested in reforestation and C sequestration. Currently, ECOMAP divides the national land base into nested, successively finer-level units including domains, divisions, provinces, sections, and subsections. Moving from coarse down to increasingly fine levels, the fundamental ecological units are defined first by broad climate zones (domains, of which there are three in CONUS), then by regional climate types, vegetation affinities and soil Orders (divisions), then by increasingly localized information about climate, lithology, geomorphology, and soil units classified to finer taxonomic levels (provinces, sections). Some locations, such as states, National Forests, and ecological reserves, have finer-level ecological unit classifications that nest sub-subsections, landtype associations, landtypes, and landtype phases into the ECOMAP hierarchy, but these are less common and culminate in more locally resolved spatial units than the results we present here. For our analyses, we retain a high-level view, exploring regional variation only down to the province level (hundreds of thousands of square km), where within-group sample sizes (e.g., hundreds to thousands of ISCN profiles or NFI plots) are sufficient to ensure that statistical tests are not influenced by lurking or confounded variables. As described in "Aboveground C stocks from FIADB", FIA datasets contained ECOMAP classifications; for ISCN sites, we used an approach similar to other overlay data types to extract ECOMAP classifications. Specifically, we downloaded domain, division, and province polygons from the USDA-Forest Service Geodata Clearinghouse (https://data.fs.usda.gov/geodata/), and used the 'extract attributes for points' tool in ArcGIS to assign each ISCN geolocation to its appropriate place in the ecoregional classification system.

Data analysis: approach and tests

Before beginning our data analyses or the data synthesis and manipulation activities described above, we defined the specific statistical tests needed to address the objectives of this work. Explicit definition of statistical tests at the beginning not only informed the structure of our datasets and approach to manipulation, but also necessitated critical consideration of strengths and limitations of the very large datasets underlying this work. Because ISCN and FIA databases contain data generated by a number of investigators working across the range of lands and ecosystems in the CONUS, their size and extent simultaneously enable and challenge far-reaching inferences. Perhaps most importantly, in very large datasets such as these, skewed distributions are to be expected. Whether due to erroneous data entry, e.g., unrealistically high C stocks for an ISCN soil profile, representing truth (e.g., a deep wetland soil with massive C stocks), or present for other reasons, right-skewed distributions were obvious for most response parameters (e.g., forest stand ages, biomass or soil C stocks) in our datasets. Rather than remove such observations as statistical outliers, or allow their magnitude to skew mean values in parametric statistics, we chose to use nonparametric tests of medians in our analyses. Specifically, for two-group comparisons, we used the Mann–Whitney U test, and for comparing the medians of three or more groups, we used Kruskal–Wallis with Dunn's multiple comparisons procedure. In addition to retaining as many observations as possible while avoiding leveraging by extreme values, we argue that this approach is actually more appropriate than parametric statistics for the scope of our analysis and its questions of interest. Thus, for the portions of our data analysis that depend upon inferential statistics, we accepted test results as significant if $P < 0.05$, and the utilized median, percentiles (25th and 75th), and interquartile range (IQR) as the basis for assessing differences in the distribution of observations within groups. For some tests, we also interpreted the Kruskal–Wallis H statistic associated with each categorical predictor as a relative ranking of its predictive strength.

For several statistical tests intended to infer the impacts of land use (cultivation, reforestation, and natural forest) on soil C and physical properties, we utilized a pedologically informed conceptual approach previously described in Nave et al. (2013), and described briefly here. In particular, we interpreted the presence of Ap horizons (sometimes called plow layers) in soil profiles as evidence of cultivation (past or present). An Ap horizon is readily recognized in a soil pit by its consistent thickness and clear abrupt boundary over underlying horizons, and may persist for decades to centuries following agricultural abandonment (Compton and Boone 2000). Most Ap horizons in our dataset were in lands categorized as cultivated by NLCD; the interpretation of these cases is self-explanatory. However, many soil profiles indicated by NLCD as having woody vegetation also had Ap horizons; we interpreted these as evidence of reforestation on previously cultivated soils. By defining a condition for a third land use group (natural forest) as a soil profile supporting woody vegetation but lacking an Ap horizon, we made statistical comparisons between actively cultivated lands, reforesting cultivated lands, and never-cultivated forests, the latter two groups including both forest and shrub/scrub land covers for reasons described in 2.1.2. Before turning to the Results, we clarify two important points regarding our treatment of land cover and use. First, by inferring that forest soils without Ap horizons were never cultivated, we may sometimes mis-categorize land use, i.e., where erosion eliminated Ap horizons before trees were established on badly degraded cultivated soils. Second, and more importantly, it is important to recognize that because our data sources possess different types of information, they must be used to address only those specific questions to which they are suited. Specifically, while we rely on indirect evidence to assess three land uses (cultivated, reforesting, natural forest) for the ISCN-NLCD observations, NFI plot data make direct assessments that offer more detail about land uses (e.g., previous forest vs. nonforest, afforestation and reforestation as different types of forest establishment). Throughout the Results and Discussion, we clearly indicate which data sources have been used in order that readers can refer to the Methods we have reported above, appreciating how the data used constrain the inferences gained.

Results

National snapshot: soil and aboveground biomass C stocks by land cover

For ISCN sites across the U.S., soils dominate AGWB as the principal ecosystem C pool for all land cover types (Table 1), and land cover types differ significantly in their median whole-profile soil C storage ($P<0.001$). Wetlands (1139 profiles) have the greatest soil C storage, followed by shrub/scrub (1743 profiles), cultivated (4568 profiles), and pasture/grass, developed, and forest cover types holding the least (n = 6089, 1483, and 7619, respectively). In contrast to their low soil profile C stocks, lands covered by forest have significantly greater median C storage in AGWB than all other land cover types ($P<0.001$); wetlands and developed lands are intermediate, while for sites with shrub/scrub, pasture/grassland, and cultivated land covers, the median AGWB is 0. In terms of their combined C stocks in the whole soil profile plus AGWB, C stocks are highest in wetlands (n = 340), intermediate to high in forest (n = 2719) and shrub/scrub (n = 565) land cover types, low to intermediate for cultivated (n = 1350) and developed (n = 593) lands, and lowest in pasture/grassland (n = 1885) cover types. Here, it is important to note that the median values for combined soil + biomass C that are reported in Table 1 are not direct sums of the independent median values of soil C and biomass C within each land cover type. This is because the median value for each of these three C stocks (soil C, biomass C, summed soil + biomass C) is actually a different observation (i.e., location). In other words, the ISCN-NLCD site that was the median in terms of its profile total C stock was not also the median site in terms of biomass C stock, nor were either of these sites the median observation in the sum of these two ecosystem pools.

Table 1 Storage of C within soil (profile total), aboveground woody biomass (AGB), and their sum, for major land cover groups in the U.S.

Pool	Developed	Cultivated	Pasture/grass	Shrub/scrub	Forest	Wetland
Soil	105 (64–187)^{d}	119 (70–205)^{c}	106 (70–175)^{d}	133 (64–290)^{b}	105 (70–185)^{d}	151 (81–291)^{a}
AGB	10 (0–39)^{b}	0 (0–1)^{c}	0 (0–10)^{c}	0 (0–12)^{c}	45 (34–58)^{a}	25 (1–44)^{b}
Sum	148 (97–241)^{c}	135 (79–251)cd	124 (87–213)^{d}	182 (97–428)^{b}	156 (119–260)^{b}	234 (142–375)^{a}

Values presented are median C stocks in Mg ha^{-1} (with 1st and 3rd quartile values in parentheses). Within each C pool, land cover groups with significantly different median C stocks ($P<0.001$) are indicated with superscripts. See "National snapshot: soil and aboveground biomass C stocks by land cover" section for the number of observations within each pool × land cover group, and notes regarding summation and presentation of medians

National to regional variation in soil C stocks

At the broadest level of the ecoregional hierarchy, domains differ significantly in their profile total C storage ($P<0.001$). Specifically, the median whole-profile soil C stock in the dry domain is 136 Mg C ha^{-1}, with 25th and 75th percentiles ranging from 69 to 293 Mg C ha^{-1}; the median profile total in the humid temperate domain is 107 Mg ha^{-1}, with an IQR of 70–183 Mg C ha^{-1}. (Parenthetically, we note that we excluded data from the small number of ISCN profiles in the humid tropical domain, which occupies extreme southern Florida). Variation in profile total soil C between domains (H = 163) is less than variation between divisions (H = 1309) or provinces (H = 2234), indicating that increasingly regionalized ecological units have increasingly different soil C stocks from one another. Among many potential factors that can explain this spatial variation in profile total C stocks, natural drainage index (H = 915) is much more important than land cover (H = 195).

Within ecoregional divisions, natural drainage index is consistently the strongest predictor of variation in profile total soil C stocks (Table 2). Land cover, which co-varies with drainage index (i.e., wetland cover types equate to poor drainage classes), is also a significant predictor of variation in profile total soil C stocks within ecoregional divisions, although there is no consistent pattern as to which land cover group has the greatest or least profile total soil C stocks. While wetlands have the greatest median soil C stocks in four divisions, forest soil C stocks are greatest in three divisions, and least in one division. In two divisions (warm temperate and temperate desert), variation in profile C stocks between montane and non-montane provinces is greater than variation attributable to drainage or land cover, highlighting the utility of province-level maps for regionalized views of representative (median) profile C stocks and their variability (Fig. 1a, b).

Impacts of land use on soil C and bulk density

At the national level, cultivated, reforested, and natural forest land uses differ in their soil C concentrations, stocks, and bulk densities. However, the direction and magnitude of these

Table 2 Sources of variation in profile total soil C stocks, by ecoregional division

Division	Province	Drainage	Land cover
Marine	$P<.001$, H = 30	$P<.001$, H = 22	**$P<.001$, H = 56**
Tropical/subtr. steppe	$P=.001$, H = 16	$P=.003$, H = 20	$P<.001$, H = 27
Prairie	$P=.656$, H < 1	**$P<.001$, H = 177**	$P<.001$, H = 134
Mediterranean	$P<.001$, H = 87	$P=.018$, H = 15	**$P<.001$, H = 102**
Temperate desert	**$P<.001$, H = 111**	$P<.001$, H = 56	$P=.002$, H = 19
Warm continental	**$P<.001$, H = 176**	$P<.001$, H = 85	$P<.001$, H = 21
Temperate steppe	$P<.001$, H = 74	**$P<.001$, H = 99**	$P<.001$, H = 23
Hot continental	$P<.001$, H = 377	**$P<.001$, H = 410**	$P<.001$, H = 143
Subtropical	$P=.005$, H = 13	**$P<.001$, H = 302**	$P<.001$, H = 117
Tropical/subtr. desert	$P=.017$, H = 19	$P=.094$, H = 8	$P=.015$, H = 14

For each of the 10 divisions, cell contents show the *P* value significance and Kruskal–Wallis H statistic for one-way tests conducted using finer-level (province) spatial variation, natural drainage class, or land cover as the categorical variable. Within each division, the bold cell indicates the most significant source of variation, assessed according to the H statistic

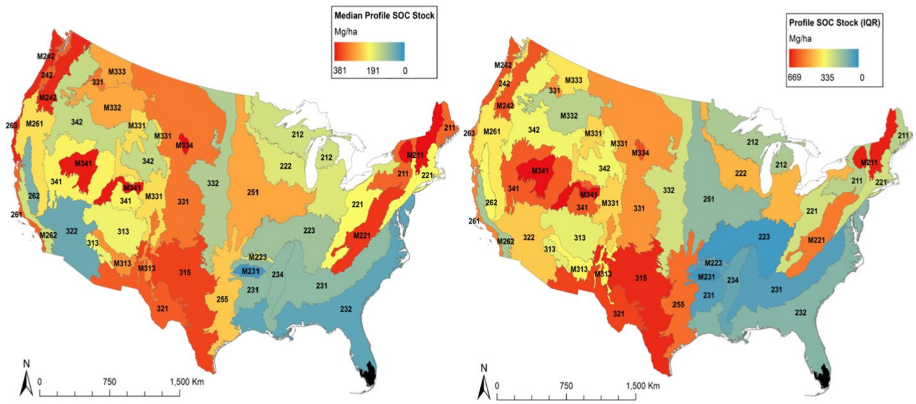

Fig. 1 Map showing the median (left panel) and interquartile range (right panel) of profile total soil carbon stocks, in Mg ha^{-1}. Warmer colors show higher (or more variable) C stocks while cooler colors show lower (or less variable) C stocks; note that the color ramp ranges differ between the two panels. Map units are the 36 ecoregional provinces delineated within the CONUS by the ECOMAP framework, less extreme southern Florida (due to low data density). (Color figure online)

differences between land uses are not consistent between topsoils (A horizons) and subsoils (B horizons). In terms of C stocks, topsoils from reforesting cultivated lands are intermediate between actively cultivated lands and natural forests; in subsoils, cultivated lands have the largest median C stocks, followed by natural forest and reforesting lands (Table 3, all $P<0.001$). Examining the properties of these soil horizons more closely, topsoils from reforesting cultivated lands have bulk densities and C concentrations intermediate between topsoils from actively cultivated lands and those from natural forests (Fig. 2; all $P<0.001$). Among subsoils (Fig. 3), bulk densities are lowest in natural forest ($P<0.001$) and similar in cultivated and reforesting soils; C concentrations are highest in natural forest, intermediate in cultivated soils, and lowest in reforesting soils ($P<0.001$).

Impacts of land use on biomass C stocks

At the national level, lands that were previously cultivated but are now reforesting have significantly lower AGWB C stocks than natural forests that, based on our inferential approach, were not previously cultivated ($P<0.001$). This result holds whether considering only ISCN sites with NLCD forest cover types, or grouping forest and shrub/scrub cover types into the combined woody vegetation cover type described in "Land cover and

Table 3 Carbon storage in Mg ha^{-1} for topsoils (A horizons) and subsoils (B horizons) of three different land uses, for soils across the CONUS

Horizon	Ongoing cultivation	Previously cultivated, reforesting	Natural forest and shrub/scrub
A	28 (15–49)^{c}	30 (19–45)^{b}	37 (21–64)^{a}
B	11 (6–22)^{a}	8 (5–15)^{c}	10 (5–23)^{b}

Values presented are median C stocks (with 25th and 75th percentiles in parentheses). Within each horizon, land uses with significantly different median C stocks ($P<0.001$) are indicated with superscripts

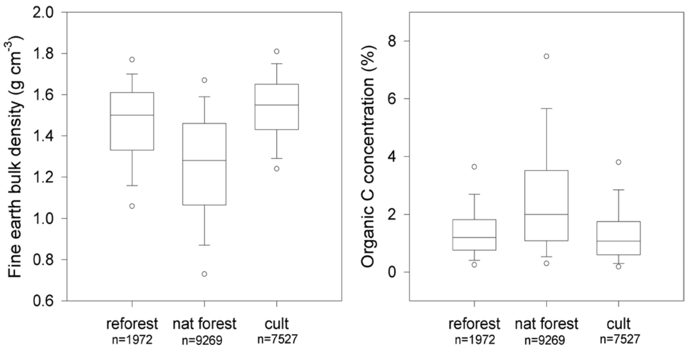

Fig. 2 Bulk density (left panel) and organic C concentration (right panel) for A horizons from soils undergoing continuous cultivation (cult), natural forest and shrub/scrub (nat forest), and previously cultivated, reforesting soils (reforest). Boxplots show medians (all groups are significantly different at $P<0.001$) and 25th and 75th percentiles; whiskers show the 10th and 90th percentiles; points are outliers (5th and 95th percentiles)

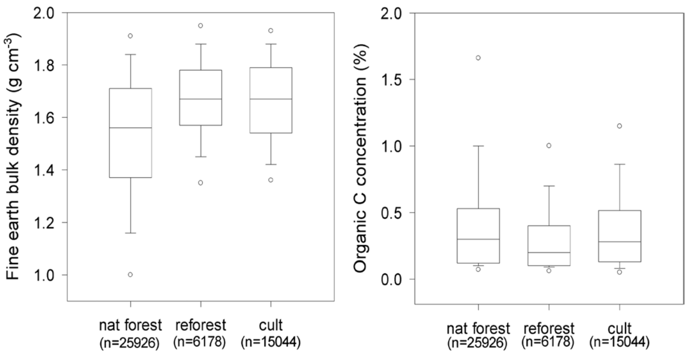

Fig. 3 Bulk density (left panel) and organic C concentration (right panel) for B horizons from soils undergoing continuous cultivation (cult), natural forest and shrub/scrub (nat forest), and previously cultivated, reforesting soils (reforest). Boxplots show medians, 25th and 75th percentiles; whiskers show the 10th and 90th percentiles; points are outliers (5th and 95th percentiles). Carbon concentrations differ significantly between all groups; bulk density is significantly lower ($P<0.001$) for natural forest from the other two land uses, which are not significantly different

biomass overlay data" section. For all woody vegetation lands collectively, median AGWB C stocks are 31 Mg C ha^{-1} (IQR = 15–45) for (previously cultivated) reforesting lands, versus 44 Mg C ha^{-1} (IQR = 27–57) for natural forest lands.

Examining the effects of forest establishment type and stand age using the more detailed, direct data from the NFI plot network reveals several significant patterns. Across the U.S., forests resulting from afforestation, reforestation, or forest establishment (planted or natural) on previously non-forested lands are younger and have lower median AGWB C stocks than naturally regenerated forests (Table 4; $P<0.001$). In terms of their median values of AGWB accumulation (C stock divided by stand age), young, deliberately established forests (afforestation and reforestation) are accumulating C in AGWB 2–3 times faster than naturally regenerated forests ($P<0.001$), except for forests on previously non-forested lands. On these lands, the rate of AGWB is roughly half that of lands maintained as forest (Table 4; $P<0.001$). When controlling for stand

Table 4 National Forest Inventory (NFI) plot data from CONUS comparing aboveground woody biomass C for two options each under three different forest conditions, including afforestation, reforestation, and forest establishment on previously non-forest land

	N	Age	Mg C ha^{-1}	Mg C ha^{-1} $year^{-1}$
Afforested				
N	23,163	65 (33–86)	43 (20–69)	0.8 (0.4–1.3)
Y	135	16 (7–35)	21 (5–63)	1.4 (0.8–2.1)
Reforested				
N	74,726	70 (43–98)	37 (13–69)	0.6 (0.2–1.1)
Y	7396	20 (10–30)	32 (10–55)	1.6 (0.9–2.5)
Prev. nonforest				
N	43,529	61 (31–83)	46 (22–71)	0.9 (0.5–1.3)
Y	1648	37 (10–71)	14 (2–43)	0.5 (0.2–1.1)

Under each condition, "N" indicates plots in which forest cover was not established by that option; "Y" indicates plots in which forest cover was established by that option. For example, for reforestation, "N" plots represent forestland not resulting from replanting (i.e., natural regeneration); "Y" plots indicate forestland that results from replanting. For each approach × option group, the table shows the number of plots, the age (years), C storage in aboveground woody biomass (Mg C ha^{-1}), and annualized rate of aboveground biomass production (Mg C ha^{-1} $year^{-1}$). Values shown are medians, with 25th and 75th percentiles in parentheses; medians are significantly different ($P<0.001$) for the N × Y comparisons within each of the three forest conditions

age to more closely examine C accumulation in AGWB over time during reforestation, planted forests accumulate more C in biomass than naturally regenerated forests (Fig. 4; $P<0.001$ for differences between medians within the first 4 time categories). Per time, the C sequestration benefit of planting is greatest in the first several decades, when AGWB C stocks are roughly three-fold greater than naturally regenerated forests. During this period, the consistently high initial stocking density of planted forests appears important to their C sequestration advantage.

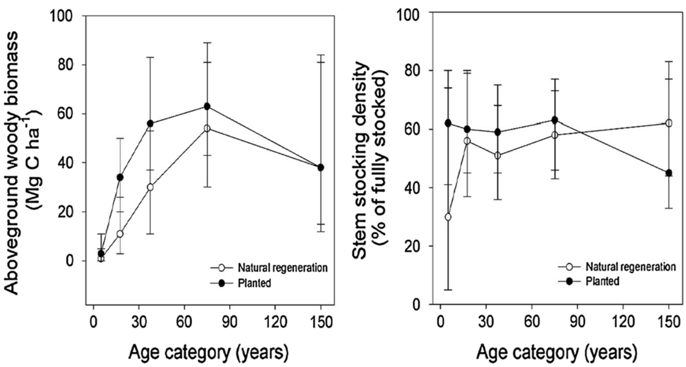

Fig. 4 Carbon storage in aboveground woody biomass (left panel) and stem stocking density (right panel) for forests resulting from natural regeneration (open symbols) versus reforestation (filled symbols). Data are from NFI plots. Points plotted are medians, which differ significantly between reforestation and natural regeneration in the first 4 time categories ($P<0.001$); error bars are the 25th and 75th percentiles

Table 5 National Forest Inventory (NFI) plot data from CONUS comparing coarse woody debris C stocks for two options each under two different forest conditions, including reforestation and forest establishment on previously non-forest land

	n	Age	Mg C ha^{-1}
Reforested			
N	20,171	70 (40–100)	3 (0–10)
Y	1871	22 (12–34)	7 (1–15)
Previous non-forest			
N	1660	55 (25–75)	1 (0–4)
Y	80	33 (8–60)	0 (0–2)

Under each approach, "N" indicates plots in which forest cover was not established by that option; "Y" indicates plots in which forest cover was established by that option. For example, for previously non-forest land, "Y" indicates forests growing on previously non-forest lands; "N" indicates forests growing on lands under continuous forest land use. For each approach × option group, the table shows the number of plots, the age (years), and C storage in coarse woody debris (Mg C ha^{-1}). Values shown are medians, with 25th and 75th percentiles in parentheses; medians are significantly different ($P < 0.001$) for the N × Y comparisons within each of the two forest conditions

Table 6 Carbon storage in coarse woody debris for forests resulting from natural regeneration versus planting (reforestation)

Age class	Natural	Planted
< 10	0 (0–4)	5 (1–14)
10–25	0 (0–1)	6 (1–14)
25–50	0 (0–3)	7 (2–18)
50–100	3 (1–9)	7 (3–17)
> 100	10 (3–21)	6 (3–16)

Values are medians, which differ significantly between reforestation versus natural regeneration in all time categories ($P < 0.001$), with 25th and 75th percentiles in parentheses

Impacts of land use on woody debris C stocks

At the national level, NFI data on CWD C stocks (Table 5) show similar trends to AGWB C stocks. First, planted forests and forests growing on previously nonforest lands are significantly younger than naturally regenerated forests and lands under continuous forest uses, respectively. Second, planted forests have median CWD C stocks approximately double those of naturally regenerated forests, and stocks of C in CWD are significantly greater in lands maintained under forest cover than in forests growing on previously nonforest land (both $P < 0.001$). In all cases, these C stocks are very small relative to the soil C and AGWB pools previously described. In terms of temporal patterns, planted forests have significantly larger CWD C stocks than naturally regenerated forests throughout the first century of forest development ($P < 0.001$). However, while CWD C stocks in planted forests appear to maintain at more or less steady state over time, naturally regenerated forests begin accumulating substantial CWD C during the decades approaching the close of the first century, and hold significantly more C in forests > 100 years old ($P < 0.001$) (Table 6).

Regional patterns in land use and a national perspective on reforestation

Regional variation in land use transitions and reforestation activities indicate that opportunities for C sequestration resulting from forest establishment are not equally distributed across the U.S. Considering ISCN sites with woody vegetation and Ap horizons (i.e., reforesting sites) reveals that, in the prairie and subtropical ecoregional divisions, over 1/3 of lands now possessing woody vegetation (forest or shrub/scrub) were once plowed (Fig. 5). Based on results pertaining to topsoil C stocks (3.3), reforesting soils in these divisions are currently recovering C lost during historic cultivation and are likely to continue doing so as long as forests are allowed to continue recovering. Additionally, NFI plot data show that the percentages of forestland less than 10 years old that result from replanting are mostly low across the U.S., but are on the order of 50% in the subtropical and marine divisions (Fig. 6). Importantly, these two divisions also have the highest median rates of AGWB C accumulation; divisions with the lowest percentage of young forests resulting from reforestation generally had the lowest median rates of AGWB C accumulation (Fig. 7).

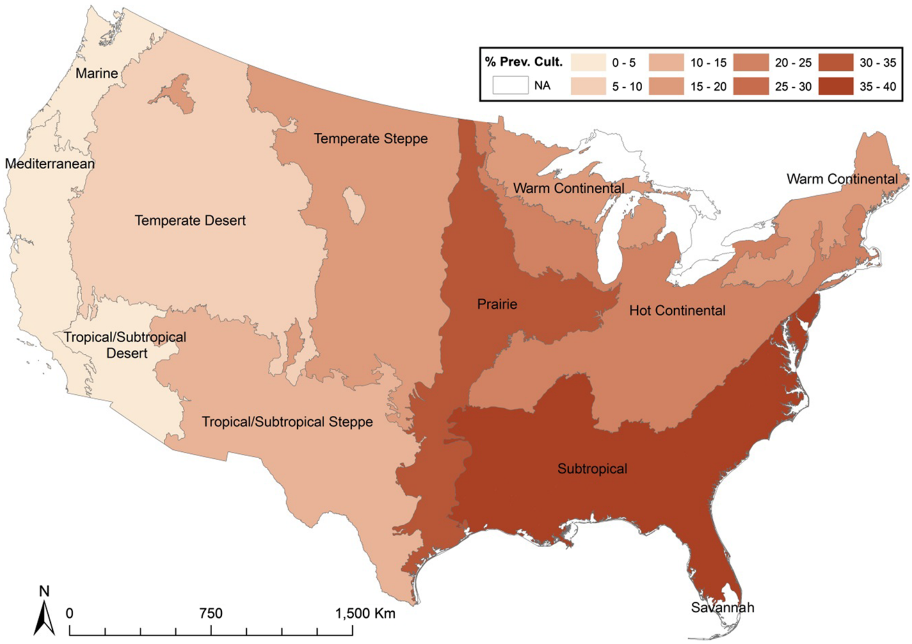

Fig. 5 Map showing the percentage of ISCN sites covered with woody vegetation that also possess an Ap horizon, indicative of past cultivation. Darker shading indicates a higher proportion of lands now covered in woody vegetation that were previously cultivated. Map units are the 10 ecoregional divisions delineated within the CONUS by the ECOMAP framework, less extreme southern Florida (Savannah division, due to low data density). (Color figure online)

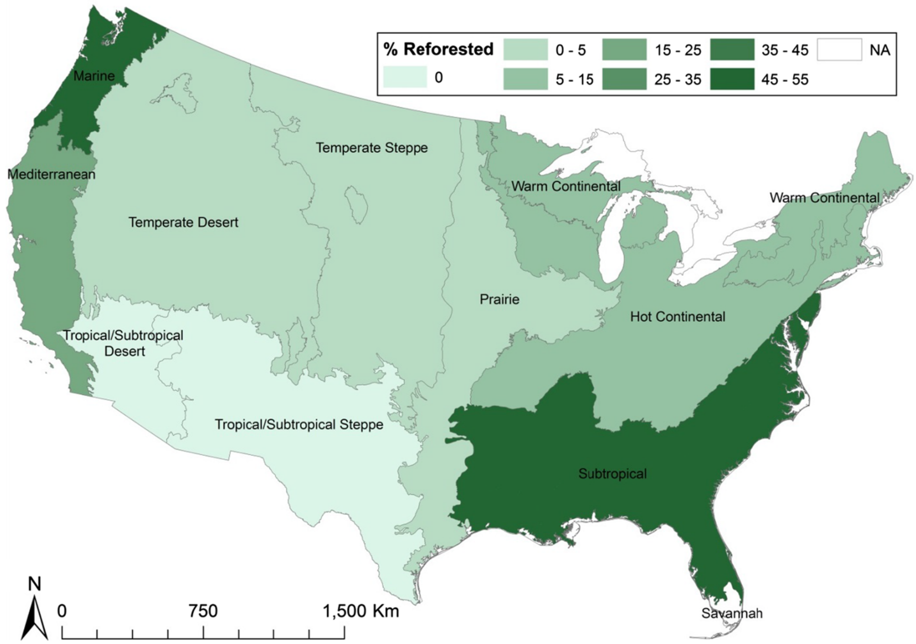

Fig. 6 Map showing the percentage of forests less than 10 years old that result from deliberate planting, based on NFI plot inventory data. Darker shading indicates a higher proportion of reforestation. Map units are the 10 ecoregional divisions delineated within the CONUS by the ECOMAP framework, less extreme southern Florida (Savannah division, due to low data density). (Color figure online)

Discussion

Key findings

The results of our analysis support four key inferences regarding land use impacts on C sequestration, discussed in subsequent sections, and highlight the large, measureable benefits that reforestation has for ecosystem C storage. First, in the bigger picture, soils are the dominant storehouse of C in all ecosystems and land cover types in the U.S., and variation in soil C pool sizes across the nation has less to do with land cover than with other factors. Nonetheless, soils hold the potential for long-term C increases following specific land use transitions; namely, where cultivated lands are converted to forest land uses. Third, the establishment of woody vegetation delivers immediate to multi-decadal C sequestration benefits in biomass and woody debris pools. Fourth, opportunities for reforestation-enhanced C sequestration (whether ongoing or not yet initiated) are not equally distributed across the U.S. Taken together, these inferences suggest that an immediate, yet phased and spatially selective approach to reforestation can enhance terrestrial C sequestration in the U.S. for decades to centuries to come.

Variation in soil and AGWB C stocks between land cover types, and the importance of ecoregional variation in these C pool sizes, does more than provide a broad overview of contemporary patterns (Tables 1, 2). More importantly, this national snapshot of C stocks by land cover suggests that the majority of the C held in terrestrial ecosystems (i.e., that in

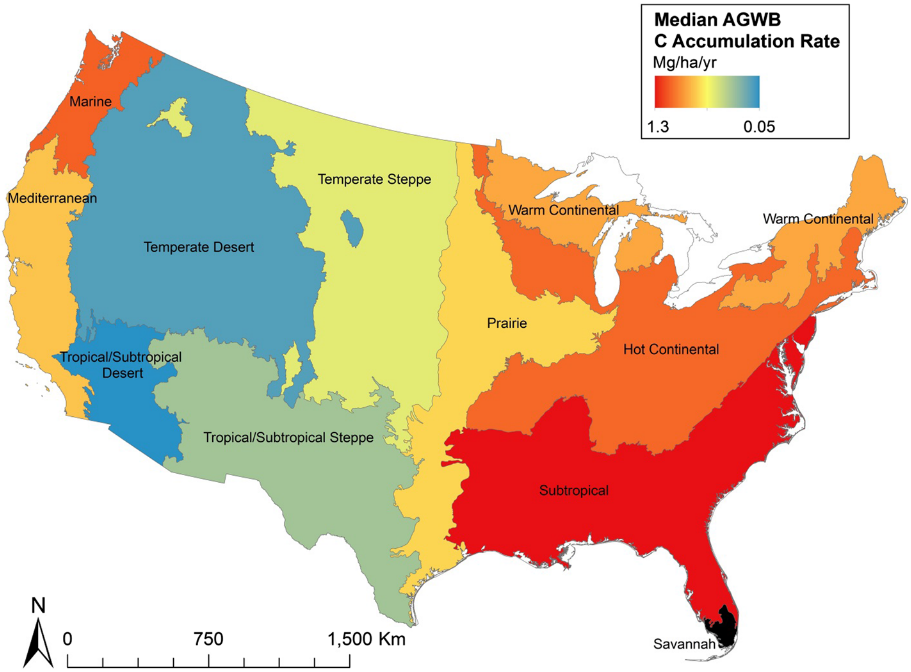

Fig. 7 Map showing the median rate of annual C accumulation in aboveground woody biomass (Mg C ha^{-1} year^{-1}). Warmer colors show higher rates while cooler colors show lower rates. Map units are the 10 ecoregional divisions delineated within the CONUS by the ECOMAP framework, less extreme southern Florida (Savannah division, due to low data density). (Color figure online)

soil) is not especially responsive to typical land use decisions. On the other hand, such a broad, observational snapshot is not a high-confidence approach to quantifying the impacts of a specific land use transition, such as reforestation on formerly cultivated soil, on ecosystem C stocks. In particular, in a nationwide assessment such as this, there is the potential for nonrandom spatial patterns in land use, such as cultivation of inherently richer soils and forest persistence on poor soils, to obscure true effects of reforestation. Furthermore, the use of remotely sensed land cover data (despite its validity according to independent observations in the ISCN DB) as a proxy for land use carries many problems of interpretation, and requires alternative sources of information to quantify C sequestration impacts resulting from reforestation.

The detailed soil descriptions and C data, coupled with remotely sensed land cover in the ISCN DB allow attribution of land use and quantification of its impacts on soil C. Specifically, the separation of cultivated lands, reforesting cultivated lands, and natural forest lands demonstrate the current status and potential for continued C sequestration during forest establishment on formerly plowed soils. Comparing median topsoil C stocks across these three land uses (Table 3) suggests that, in general, deforestation and cultivation release 25% of topsoil C stocks relative to a forested baseline (28 vs. 37 Mg C ha^{-1}). Given a median topsoil C stock of 30 Mg C ha^{-1} in reforesting cultivated lands, it appears these soils have yet to recover the majority of their "lost" C, assuming that never-cultivated natural forests represent an attainable long-term target. Considered collectively with results from subsoil horizons, which have smaller C stocks but show a net decrease in soil

C during reforestation compared to natural forest or cultivated lands, these results fit with patterns observed during long-term studies of individual sites undergoing post-agricultural reforestation. For example, on the Calhoun Experimental Forest (South Carolina), which suffered severe soil degradation during cultivation in the 19th and early 20th centuries, reforestation by *Pinus taeda* L. since the 1950s has been driving net accumulation of C in the topsoil and net loss of C from subsoils (Mobley et al. 2015; Richter et al. 1999). More broadly, quantitative reviews have demonstrated that while turnover and net replacement of deep soil C such as this is typical during forest regrowth, whole-profile C stocks typically increase during reforestation (Guo and Gifford 2002; Laganiere et al. 2010; Nave et al. 2013; Post and Kwon 2000). In general, residence times of soil C increase, and rates of C cycling processes decrease, with depth (Heckman et al. 2014; Schrumpf et al. 2013; von Lutzow et al. 2006). Given that even relatively "fast-cycling" soil horizons, such as topsoils, have C residence times spanning many decades to centuries, with deeper horizons holding C that turns over on century- to millennial scales, the recovery times for soil C lost during cultivation are likely quite long. Therefore, a sustained commitment to reforestation, rather than re-initiation of cultivation, is a requirement for meaningful C gains in reforesting soils. This is all the more important given predicted increases land use transfers from forest to non-forest land uses as the twentyfirst century proceeds (USDA-Forest Service 2016), as these would sacrifice soil C gains in reforesting soils on a long-term trajectory to C recovery.

In the immediate to medium-term, such as the multi-decadal period over which most forests are allowed to mature before harvesting in the U.S., AGWB is the pool that presents a clear opportunity for reforestation to enhance terrestrial C sequestration. While NFI data allowing an assessment of reforestation are much more abundant than data allowing for assessment of afforestation (Table 4), both land use decisions show the same patterns relative to naturally regenerated forests. Specifically, that planted forests tend to be younger and faster-growing than naturally regenerated forests. Generalizations that forest biomass accumulation rates are highest in younger stands (Gower et al. 1996; Ryan et al. 1997) argue for making direct comparisons of planted versus naturally regenerated forests within specific stand age ranges, yet even when age differences are controlled in this way the benefit of planting remains clear, at least through the first several decades (Fig. 4). By appearances, the key to C sequestration enhancements in planted versus naturally regenerated forests is the high initial stocking density of planted stands (Sample 2017). However, the leveling off of stocking density and AGWB C stocks in the latter decades of the first century (and the decline in both beyond 100 years of development) highlights the importance of accounting for biomass removals or stand-eliminating disturbances (e.g., fires) in the life cycle C budgets of mid- to later-successional forests. In other words, net declines in AGWB in forests > 100 years old indicate loss of woody C from the ecosystem; whether this material is lost fairly quickly to the atmosphere (e.g., due to fire or bioenergy combustion) or sequestered in a long-lived pool such as construction materials has a major impact on the broader role of forests in the C cycle (Brunet-Navarro et al. 2016; Heath et al. 2011; Smyth et al. 2014).

To the degree that they provide independent assessments of the same land use transition (forest establishment on previously nonforested land), the combined ISCN soil profile + NLCD + NBCD2000 overlay data ("Impacts of land use on biomass C stocks" section) and the FIADB plot data (Table 4) generate mutually consistent results. Specifically, both approaches suggest slower AGWB C accumulation in forests growing on previously non-forested lands. As many of these lands were likely cultivated in the past, these results suggest that reforestation of agricultural lands may be preferentially occurring on lands that have been degraded, perhaps through the deterioration of soil properties that support tree growth, such as lower organic matter and higher bulk density (Figs. 2, 3). Regardless the mechanism, the slower biomass accumulation rates on previously nonforest (cultivated) lands illustrate how targeted reforestation of such lands can fit into a phased approach for maximizing C sequestration at a national level. Specifically, reforestation on degraded agricultural lands represents a land use transition with longer-term returns. Whereas there is a clear and immediate C accrual benefit of immediately replanting forests that have been recently disturbed or harvested (Table 4, Fig. 4), with the largest gains above natural regeneration during the first 1–3 decades, the slower recovery time for soil characteristics and forest production rates suggests that reforestation on depleted agricultural soils may play a role more in the 50 to > 100 year timeframe as soil quality begins to improve.

Patterns of convergence and disparity between reforestation activity, forest growth rates, and the establishment of forests on previously cultivated lands point the way to a range of priorities and opportunities for increasing C sequestration through tree planting in the U.S. (Figs. 5, 6, 7). Perhaps most importantly, the generally low rates of replanting across the Nation indicate that any investment in reforestation can improve the situation from its current, chronically under-attaining level. And, because AGWB C accumulation rates differ so widely across ecoregional divisions, it is apparent that while some ecoregions should not be prioritized for large-scale increases in reforestation (e.g., the dry tropical/subtropical divisions of the interior Southwest), even marginal increases in high-productivity divisions (e.g., marine in the Northwest and subtropical in the Southeast) can produce large C gains. At a minimum, these gains include C sequestered in AGWB, while in the Southeast, reforestation is also adding significant C to historically cultivated soils that recover and hold C over longer timescales. In other areas, such as the warm and hot continental divisions of the Northeast, AGWB accumulation rates are moderately high, yet reforestation rates scarcely exceed 10% of forests less than 10 years old. Increased reforestation here—especially in the hot continental division, where historic cultivation was quite extensive—has the potential to make a large impact on the national forest sector C balance, especially given the large land area. The prairie and temperate steppe divisions of the central U.S. furnish a final example. Here, many lands currently covered by woody vegetation were once cultivated, yet very few of these lands originate from deliberate reforestation (or afforestation if they were truly never forested). Given that these lands are currently realizing C accruals in soils due to the establishment of woody vegetation, a targeted increase in tree planting in this region, rather than passive woody encroachment following agricultural abandonment, can likely increase C sequestration over longer timeframes. Similar efforts have been mounted in the past, such as during the U.S. dust bowl era of the 1930s, when over one million hectares of National Forest System lands were planted or seeded by the Civilian Conservation Corps, and elsewhere, such as in degraded sand and loess soil regions of China (Liu et al. 2008) more recently for similar reasons.

Caveats and considerations

In this paper, we have referred variously to land cover and to land use, in the interest of speaking explicitly to the land attribute in question. Our principal aim in this analysis is focused on land *use*—the activity being conducted on a parcel of land; most particularly on forest establishment (whether through deliberate planting or natural regeneration, afforestation or woody encroachment). However, in many cases, we have relied upon remotely-sensed land *cover* data as an indication of land use, and in these cases we use the term land cover to be specific while acknowledging that a snapshot of land cover does not necessarily indicate land use. For example, a freshly clearcut forest could be detected as a land cover of shrub/scrub based on the low stature and density of its woody vegetation, even though the actual land use was forest remaining forest, its temporary disturbance aside. On the other hand, to accurately attribute land *use*, additional information collected via on-the-ground observation is necessary. For cases in which we have such information, such as through the combination of soil morphology (Ap horizon presence/absence) and land cover (cultivated vs. covered by woody vegetation), we have used the term land use with confidence. Others have addressed the issue of land use versus land cover in the context of large-scale land assessments (e.g., Coulston et al. 2014; Woodall et al. 2016); here, our intent is to highlight the potential limitation of our inferences resulting from reliance on remote sensing data. In the end, corroborative results derived from ISCN land cover overlay data and data from NFI plots ("Impacts of land use on biomass C stocks" section) allows our inferences to speak for themselves, especially in light of other limitations to our approach.

The most important caveats that must be considered in this analysis pertain to our use of data collected across space and time as a means to make indirect comparisons of land use. First, the past—in this case, currently available inventory data that reflect recent land cover, land use and management practices—may not predict the future. In that regard, inferences that are forward-looking, such as the potential for C accumulation to continue on lands that have undergone cultivation-to-forest transition, are open to question. Second, because we rely for many of our inferences on space-for-time substitutions, such as NFI plots spanning a range of forest ages, there is the potential for our approach to mis-attribute causation or obscure important underlying constraints. For example, it is possible that certain agricultural lands are preferentially abandoned for underlying factors that later influence the rate of forest biomass accumulation, and it is these factors (rather than cultivation itself) that results in slower growth rates for forests on previously cultivated lands. Similarly, it is possible that the soil datasets contributed to ISCN comprise a non-representative sample of lands in the U.S., and this could obfuscate trends that we do, or do not detect and report in this analysis.

A third consideration that could impact our results pertains to those C pools that we did versus did not include in our analyses. Specifically, in this paper, we do not report the contributions of trees < 12.7 cm diameter, or of roots (coarse or fine) to ecosystem C stocks. Early in analyses, we examined data from NFI plots, and upon determining that small trees represent < 10% of the AGWB on > 90% of the plots, chose to exclude these as a pool of interest. Roots- in particular, the coarse, woody roots that represent a C pool that is similarly long-lived to AGWB, are likely a significant C stock at all spatial levels (plots, ecosystems, ecoregions). However, the NFI approach, similar to that often used in large-scale C work, is to estimate the pool size of this belowground woody biomass as a static fraction of AGWB (e.g., 20%), and estimate it on that basis. Rather than inflate our C stock analyses by including these uncertain estimates, we exclude them. While the overall result is likely that we underestimate C sequestration due to reforestion as a consequence, we suggest this is an acceptable trade-off in an analysis that otherwise incorporates so many sources of uncertainty. Ultimately, while there are still other caveats and considerations that could be raised around this work, its inferences are based on very large datasets that provide a degree of confidence in its overarching results, and we offer its results as self-supporting.

Conclusions

Wide-ranging data from independent, complementary sources suggest that reforestation enhances C sequestration in multiple ecosystem pools, differing in their C residence times, at regional to national levels. In general, soil C stocks are not particularly sensitive to land cover type, but specific land use transitions, such as the establishment of forests on formerly cultivated lands, causes increases in topsoil C storage. Under these situations, rates of C accumulation in aboveground woody biomass are lower than rates observed for continuous forest land uses, but represent an additional pool for C gains during reforestation. In forest lands that have been harvested or affected by stand-eliminating disturbances, deliberate re-planting realizes two- to three-fold gains in C accumulation in aboveground woody biomass compared to natural regeneration. Coarse woody debris C stocks, while much smaller overall, are also increased as a result of reforestation. Given wide variation in fundamental ecologic factors, such as climate and geology, that influence forest growth rates, an ecoregional framework is well-suited to identifying and prioritizing areas for reforestation efforts at regional to national levels.

Acknowledgements We thank organizers Kasten Dumroese, Nicole Balloffet, and Jim Vose, and the participants of the Reforestation Matters workshop in Portland, OR, 12–13 April 2017, for the opportunity to contribute this synthesis to the reforestation science effort in the U.S. We are grateful to the USDA-Forest Service, Northern Research Station (Agreements No. 13-CR112306-077, 16-CR-112306-071, and 17-CR-11242306-028) and the National Science Foundation (Award No. EF-1340681) for the financial support to conduct this analysis. Lastly, we are grateful for the reviews provided by two anonymous referees and the Guest Associate Editor, who have helped to improve this work.

References

Bechtold WA, Patterson PL (eds) (2005) The enhanced forest inventory and analysis program—national sampling design and estimation procedures. General technical report SRS-80. U.S. Department of Agriculture, Forest Service, Southern Research Station, Asheville, NC, 85 p

Bentz BJ, Regniere J, Fettig CJ, Hansen EM, Hayes JL, Hicke JA, Kelsey RG, Negron JF, Seybold SJ (2010) Climate change and bark beetles of the western United States and Canada: direct and indirect effects. Bioscience 60:602–613

Birdsey R, Pregitzer K, Lucier A (2006) Forest carbon management in the United States: 1600–2100. J Environ Qual 35:1461–1469

Bond-Lamberty B, Wang CK, Gower ST (2004) Net primary production and net ecosystem production of a boreal black spruce wildfire chronosequence. Glob Change Biol 10:473–487

Brunet-Navarro P, Jochheim H, Muys B (2016) Modelling carbon stocks and fluxes in the wood product sector: a comparative review. Glob Change Biol 22:2555–2569

Buell GR, Markewich HW (2004) Data compilation, synthesis, and calculations used for organic-carbon storage and inventory estimates for mineral soils of the Mississippi River basin. US geological survey professional paper 1686-A, U.S. Department of the Interior, U.S. Geological Survey, Reston, VA

Caspersen JP, Pacala SW, Jenkins JC, Hurtt GC, Moorcroft PR, Birdsey RA (2000) Contributions of land-use history to carbon accumulation in US forests. Science 290:1148–1151

Cleland DT, Avers PE, McNab WH, Jensen ME, Bailey RG, King T, Russell WE (1997) National hierarchical framework of ecological units. In: Boyce MS, Haney A (eds) Ecosystem management: applications for sustainable forest and wildlife resources. Yale University Press, New Haven, pp 181–200

Cole JA, Johnson KD, Birdsey RA, Pan Y, Wayson CA, McCollough K, Hoover CM, Hollinger DY, Bradford JB, Ryan MG, Kolka RK, Weishampel P, Clark KL, Skowronski NS, Hom J, Ollinger SV, McNulty SG, Gavazzi MJ (2013) Database for landscape-scale carbon monitoring sites. General technical report, U.S. Department of Agriculture, Forest Service, Northern Research Station. GTR-NRS-119

Compton JE, Boone RD (2000) Long-term impacts of agriculture on soil carbon and nitrogen in New England Forests. Ecology 81:2314–2330
Coulston JW, Reams GA, Wear DN, Brewer CK (2014) An analysis of forest land use, forest land cover and change at policy-relevant scales. Forestry 87:267–276
Coulston JW, Wear DN, Vose JM (2015) Complex forest dynamics indicate potential for slowing carbon accumulation in the southeastern United States. Scientific reports 5
Creutzburg MK, Scheller RM, Lucash MS, LeDuc SD, Johnson MG (2017) Forest management scenarios in a changing climate: trade-offs between carbon, timber, and old forest. Ecol Appl 27:503–518
Dumroese RK, Williams MI, Stanturf JA, St. Clair JB (2015) Considerations for restoring temperate forests of tomorrow: forest restoration, assisted migration, and bioengineering. New Forest 46:947–964
Fry J, Xian G, Jin S, Dewitz J, Homer C, Yang L, Barnes C, Herold N, Wickham J (2011) Completion of the 2006 national land cover database for the conterminous United States. Photogramm Eng Remote Sensing 77:858–864
Gough CM, Vogel CS, Harrold KH, George K, Curtis PS (2007) The legacy of harvest and fire on ecosystem carbon storage in a north temperate forest. Glob Change Biol 13:1935–1949
Gower ST, McMurtrie RE, Murty D (1996) Aboveground net primary production decline with stand age: potential causes. Trends Ecol Evol 11:378–382
Guo LB, Gifford RM (2002) Soil carbon stocks and land use change: a meta analysis. Glob Change Biol 8:345–360
Heath LS, Smith JE, Skog KE, Nowak DJ, Woodall CW (2011) Managed Forest Carbon Estimates for the US Greenhouse Gas Inventory, 1990-2008. J Forest 109:167–173
Heckman K, Welty-Bernard A, Rasmussen C, Schwartz E (2009) Geologic controls of soil carbon cycling and microbial dynamics in temperate forests. Chem Geol 267:12–23
Heckman KA, Campbell JL, Powers H, Law B, Swanston C (2013) The influence of fire on the radiocarbon signature and character of soil organic matter in the Siskiyou Forest, Oregon. Fire Ecol 9:40–56
Heckman K, Throckmorton H, Clingensmith C, Vila FJG, Horwath WR, Knicker H, Rasmussen C (2014) Factors affecting the molecular structure and mean residence time of occluded organics in a lithosequence of soils under ponderosa pine. Soil Biol Biochem 77:1–11
Hicke JA, Allen CD, Desai AR, Dietze MC, Hall RJ, Hogg EH, Kashian DM, Moore D, Raffa KF, Sturrock RN, Vogelmann J (2012) Effects of biotic disturbances on forest carbon cycling in the United States and Canada. Glob Change Biol 18:7–34
Homer CC, Huang L, Yang B Wylie, Coan M (2004) Development of a 2001 National Landcover Database for the United States. Photogramm Eng Remote Sensing 70:829–840
Homer CG, Dewitz JA, Yang L, Jin S, Danielson P, Xian G, Coulston J, Herold ND, Wickham JD, Megown K (2015) Completion of the 2011 National Land Cover Database for the conterminous United States-representing a decade of land cover change information. Photogramm Eng Remote Sensing 81:345–354
Jin WC, He HS, Thompson FR, Wang WJ, Fraser JS, Shifley SR, Hanberry BB, Dijak WD (2017) Future forest aboveground carbon dynamics in the central United States: the importance of forest demographic processes. Scientific reports 7
Kashian DM, Romme WH, Tinker DB, Turner MG, Ryan MG (2006) Carbon storage on landscapes with stand-replacing fires. Bioscience 56:598–606
Kellndorfer J, Walker W, Kirsch K, Fiske G, Bishop J, LaPoint L, Hoppus M, Westfall J (2013) NACP aboveground biomass and carbon baseline data, V. 2 (NBCD 2000), U.S.A., 2000. Data set. http://daac.ornl.gov from ORNL DAAC, Oak Ridge, Tennessee, USA
Kurz WA, Dymond CC, Stinson G, Rampley GJ, Neilson ET, Carroll AL, Ebata T, Safranyik L (2008) Mountain pine beetle and forest carbon feedback to climate change. Nature 452:987–990
Laganiere J, Angers DA, Pare D (2010) Carbon accumulation in agricultural soils after afforestation: a meta-analysis. Glob Change Biol 16:439–453
Law BE, Sun OJ, Campbell J, Van Tuyl S, Thornton PE (2003) Changes in carbon storage and fluxes in a chronosequence of ponderosa pine. Glob Change Biol 9:510–524
Liang S, Hurteau MD, Westerling AL (2017) Response of Sierra Nevada forests to projected climate-wildfire interactions. Glob Change Biol 23:2016–2030
Liu JG, Li SX, Ouyang ZY, Tam C, Chen XD (2008) Ecological and socioeconomic effects of China's policies for ecosystem services. Proc Natl Acad Sci USA 105:9477–9482
MacDonald SE, Landhausser SM, Skousen J, Franklin J, Frouz J, Hall S, Jacobs DF, Quideau S (2015) Forest restoration following surface mining disturbance: challenges and solutions. New Forest 46:703–732
McNab WH, Cleland DT, Freeouf JA, Keys JE, Nowacki GJ, Carpenter CA (2007) Description of ecological subregions: sections of the conterminous United States. USDA, Forest Service, Washington, p 80

Mishra U, Riley WJ (2015) Scaling impacts on environmental controls and spatial heterogeneity of soil organic carbon stocks. Biogeosciences 12:3993–4004

Mobley ML, Lajtha K, Kramer MG, Bacon AR, Heine PR, Richter DD (2015) Surficial gains and subsoil losses of soil carbon and nitrogen during secondary forest development. Glob Change Biol 21:986–996

Nave LE, Swanston CW, Mishra U, Nadelhoffer KJ (2013) Afforestation effects on soil carbon storage in the United States: a synthesis. Soil Sci Soc Am J 77:1035–1047

Nave L, Johnson K, van Ingen C, Agarwal D, Humphrey M, Beekwilder N (2017) International Soil Carbon Network (ISCN) Database, International Soil Carbon Network (ISCN) Database, Version 3. International Soil Carbon Network. https://doi.org/10.17040/ISCN/1305039

Nave LE, Domke GM, Hofmeister KL, Mishra U, Perry CH, Walters BF, Swanston CW (2018) Reforestation can sequester two petagrams of carbon in U.S. topsoils in a century. In: Proceedings of the National Academy of Sciences of the United States. https://doi.org/10.1073/pnas.1719685115

Oswalt SN, Smith WB, Miles PD, Pugh SA (2014) Forest resources of the United States, 2012: a technical document supporting the Forest Service 2015 update of the RPA assessment. General technical report GTR WO-91, U.S. Department of Agriculture, Forest Service, Washington Office, Washington, DC, 218 pp

Post WM, Kwon KC (2000) Soil carbon sequestration and land-use change: processes and potential. Glob Change Biol 6:317–327

Puhlick J, Woodall C, Weiskittel A (2017) Implications of land-use change on forest carbon stocks in the eastern United States. Environ Res Lett 12:024011

Richter DD, Markewitz D, Trumbore SE, Wells CG (1999) Rapid accumulation and turnover of soil carbon in a re-establishing forest. Nature 400:56–58

Ryan MG, Binkley D, Fownes JH (1997) Age-related decline in forest productivity: pattern and process. In: Begon M, Fitter AH (eds) Advances in ecological research, vol 27. Elsevier Academic Press, London, pp 213–262

Sample VA (2017) Potential for additional carbon sequestration through regeneration of nonstocked forest land in the United States. J Forest 115:309–318

Schoennagel T, Balch JK, Brenkert-Smith H, Dennison PE, Harvey BJ, Krawchuk MA, Mietkiewicz N, Morgan P, Moritz MA, Rasker R, Turner MG, Whitlock C (2017) Adapt to more wildfire in western North American forests as climate changes. Proc Natl Acad Sci USA 114:4582–4590

Schrumpf M, Kaiser K, Guggenberger G, Persson T, Kogel-Knabner I, Schulze ED (2013) Storage and stability of organic carbon in soils as related to depth, occlusion within aggregates, and attachment to minerals. Biogeosciences 10(3):1675–1691

Sequeira CH, Wills SA, Seybold CA, West LT (2014) Predicting soil bulk density for incomplete databases. Geoderma 213:64–73

Smyth CE, Stinson G, Neilson E, Lempriere TC, Hafer M, Rampley GJ, Kurz WA (2014) Quantifying the biophysical climate change mitigation potential of Canada's forest sector. Biogeosciences 11:3515–3529

USDA Forest Service (2016) Future of America's forests and rangelands: update to the 2010 resources planning act assessment. General technical report WO-GTR-94, Washington, DC, 250 pp

Vogelmann JE, Howard SM, Yang L, Larson CR, Wylie BK, Van Driel JN (2001) Completion of the 1990s National Land Cover Data Set for the conterminous United States. Photogramm Eng Remote Sensing 67:650–662

von Lutzow M, Kogel-Knabner I, Ekschmitt K, Matzner E, Guggenberger G, Marschner B, Flessa H (2006) Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions—a review. Eur J Soil Sci 57(4):426–445

Watrud E, Zensen F, Darbyshire R (2012) Laws affecting reforestation on USDA Forest Service lands. Tree Plant Notes 55:39–42

Wear DN, Coulston JW (2015) From sink to source: regional variation in US forest carbon futures. Scientific reports 5

Williams CA, Collatz GJ, Masek J, Goward SN (2012) Carbon consequences of forest disturbance and recovery across the conterminous United States. Global Biogeochem Cycles 26:GB1005

Williams CA, Collatz GJ, Masek J, Huang CQ, Goward SN (2014) Impacts of disturbance history on forest carbon stocks and fluxes: merging satellite disturbance mapping with forest inventory data in a carbon cycle model framework. Remote Sens Environ 151:57–71

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

Woodall CW, Walters BF, Russell MB, Coulston JW, Domke GM, D'Amato AW, Sowers PA (2016) A tale of two forest carbon assessments in the eastern United States: forest use versus cover as a metric of change. Ecosystems 19:1401–1417

Yang S, Mountrakis G (2017) Forest dynamics in the US indicate disproportionate attrition in western forests, rural areas and public lands. PLoS ONE 12:e0171383

Zhang F, Chen JM, Pan Y, Birdsey RA, Shen S, Ju W, He L (2012) Attributing carbon changes in conterminous U.S. forests to disturbance and non-disturbance factors from 1901–2010. J Geophys Res 117:G02021

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