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Disturbance and management effects on forest soil organic carbon stocks in the Pacific Northwest

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Abstract

Carbon (C)-informed forest management requires understanding how disturbance and management influence soil organic carbon (SOC) stocks at scales relevant to landowners, forest policy and management professionals. The continued growth of datasets and publications allows powerful synthesis approaches to be applied to such questions at increasingly fine scales. Here, we report results from a synthesis that used meta-analysis of published studies and two large observational databases to quantify disturbance and management impacts on SOC stocks. We conducted this, the third in a series of ecoregional SOC assessments, for the Pacific Northwest, which comprises ~8% of the land area but ~12% of the U.S. forest sector C sink. At the ecoregional level, our analysis indicated that fundamental patterns of vegetation, climate, and topography are far more important controls on SOC stocks than land use history, disturbance or management. However, the same patterns suggested that increased warming, drying, wildland fire, and forest regeneration failure pose significant risks to SOC stocks across the region. Detailed meta-analysis results indicated that wildfires diminished SOC stocks throughout the soil profile, while prescribed fire only influenced surface organic materials and harvesting had no significant overall impact on SOC. Independent observational data corroborated the negative influence of fire on SOC derived from meta-analysis, suggested that harvest impacts may vary sub-regionally with climate or vegetation, and revealed that forests with agricultural uses (e.g., grazing) or legacies (e.g., cultivation) had smaller SOC stocks. We also quantified effects of a range of common forest management practices having either positive (organic amendments, nitrogen (N)-fixing vegetation establishment, inorganic N fertilization) or no overall effects on SOC (other inorganic fertilizers, urea fertilization, competition suppression through herbicides). In order to maximize the management applications of our results, we qualified them with ratings of confidence based on degree of support across approaches. Lastly, similar to earlier published assessments from other ecoregions, we supplemented our quantitative synthesis results with a literature review to arrive at a concise set of tactics for adapting management operations to site-specific criteria.

Key words: best management practices, carbon management, fire, fuel reduction, harvest, meta-analysis

1. Introduction

Soil organic matter (SOM) is a cornerstone of agricultural and forest productivity (Vance 2000). In soils, SOM and the organic carbon (SOC) that comprises it are vital to biogeochemical, hydrologic, and other ecosystem services that are foundational to ecosystems themselves, and the fiber, fuel, and food resources that they provide humanity (Nave et al. 2019a). Recognizing that SOC and SOM play key roles within the ecosystem and in larger issues such as the mitigation of atmospheric CO_2 pollution and climate change, many stakeholders are concerned with the potential for land use and forest management to impact SOC and SOM (Harden et al. 2018).

Broad reviews report that land use and forest management affect SOC (e.g., Certini 2005; James and Harrison 2016; Jandl et al. 2007; Post and Kwon 2000; Smith et al. 2016), and have evolved to the point that they can now review other reviews (Dignac et al. 2017; Mayer et al. 2020). This maturation of SOC management syntheses provides foundations for general understanding, and has quantified SOC impacts and their uncertainties for specific land sector activities at broad scales (Laganiere et al. 2010; Lorenz and Lal 2014; Nave et al. 2010; 2011; Thiffault et al. 2011). The value of these SOC management syntheses and the generalizations they have produced is considerable. However, the papers that have generated these foundations of our understanding share one problematic finding: they recognize that place matters, at some scale between broad synthesis and site-specific study. There are clear exceptions to many generalized rules, and even strong generalizations can be irrelevant, inaccurate, or out of context when applied to a specific ecoregion, landscape, or project. There is thus a need to apply the synthesis tools that effectively address questions of SOC management broadly at scales relevant to targeted decision making by landowners, forest managers, and policy makers.

It is now possible to use synthesis techniques to address SOC management at intermediate, if not localized scales. This possibility exists because of ongoing increases in data availability and the flexibility of the techniques themselves. For example, meta-analysis quantifies major treatment effects by synthesizing across individual studies, while using minor differences within and between studies to reveal key sources of variation in those effects (Hedges et al. 1999). However, even large meta-analyses are constrained by the origins of the studies they synthesize, making them good for identifying patterns at select sites, but unable to address the diversity of conditions across intervening spaces (Gurevitch et al. 2001). Recognizing this limitation, meta-analysis can be validated or contextualized with observations collected more widely, such as through soil survey or forest inventory programs. Such observational datasets lack experimental control, may not possess all ancillary variables, and incorporate variation that may obscure or confound treatments of interest. Nonetheless, these datasets enable comparisons and inferences over those intervening areas that have not been reported in the literature, and furthermore, ancillary variables can be obtained from other sources to create datasets that complement meta-analysis in scale, scope, and approach. This combination of approaches has proven useful in moving from broad patterns (e.g., Nave et al. 2010; 2018) to the specific soils, landscapes, and land use and management regimes of several distinct ecoregions (Nave et al. 2019b; 2021), and holds the potential to produce more nuanced applications in still more.

Forests of the Pacific Northwest are exceptional in many regards. Their biodiversity, topographic and climatic variation, stature of long-lived dominant tree species, and the formidable C-sequestering capacity of their volcanic and sedimentary derived soils lend them significance in excess of their extent. These highly recognizable aspects of forests in the Pacific Northwest also translate to nationally, if not globally significant C stocks. The forests of Oregon and Washington alone represent 7.5% of the forestland area, but 11.3% of forest C in the conterminous U.S. (CONUS). On an annual basis, forests of Oregon and Washington comprise about 12% of the annual U.S. forest sector C sink, which overall offsets 11% of annual U.S. greenhouse gas emissions (Domke et al. 2018; USDA 2020a; 2020b). However, neither the C stocks nor the C sink strength of these forests are static. Climate change, wildfire and insect disturbances, and management legacies are interacting in ways that threaten to reduce the C sink strength, or even turn the region into a C source by the middle of the 21st century (Duan et al. 2016; Kurz et al. 2008a; 2008b, Wear and Coulston 2015). This has implications for the U.S. and global forest sector C balance, regional forest health, and for the 12 million people who live amidst these forests, their health and the ecosystem services that support them (Burke et al. 2021; Siedl et al. 2016).

As impressive as they are, the statuesque trees of Pacific Northwest forests are not the largest C pool in these ecosystems. Soils to a depth of 1 m hold 44% of the forest C in Oregon and Washington, compared to 33% in aboveground biomass (USDA 2020a; 2020b). Moreover, despite the ability of dominant tree species in the region to grow and accumulate C for several centuries, much of the C in soils is stored for even longer timescales (many centuries to millennia) (Crow et al. 2007; Heckman et al. 2013; Homann et al. 2005; Smithwick et al. 2002). The loss of soil C, which is often stored for longer and slower to recover, thus has direct consequences for forest C budgets and potential negative feedbacks to ecosystem services that depend upon the maintenance of soil organic matter, such as forest productivity.

Land use, disturbances, or management can affect SOC stocks through a range of mechanisms. Most directly, the oxidation of SOC by fire or the physical disruption of soil structure that protects SOM from decomposition result in the emission of SOC to the atmosphere as CO_2 (Six et al. 2002; von Lutzow et al. 2006). Physical disruption can occur when soils are mixed, compacted, or displaced by tillage or mechanized operations, or in the case of fire, when soil heating is sufficient to eliminate SOM from structural elements such as aggregates (Bormann et al. 2008; DeGryze et al. 2004; Shabaga et al. 2017; Six et al. 2000). These mechanisms are likely to cause the largest-magnitude effects, owing to their direct action. However, these direct effects can lead to sustained, indirect SOC losses through wind or water erosion, especially for cultivated, burned, or severely harvest impacted soils that lack litter or vegetative cover (Certini 2005; Jurgensen et al. 1997; McLauchlan 2006). Other indirect mechanisms for SOC loss include: 1) diminished organic matter inputs, e.g., through delayed or failed regeneration after tree mortality, agricultural or forest harvest removals; 2) increased soil temperature and moisture that stimulate decomposition, e.g., through loss of shading or litter cover; 3) biogeochemical mechanisms, e.g., pH changes that increase microbial activity or incorporation of labile C into previously stable SOM (Adkins et al. 2020; Andersson and Nilsson et al. 2001; Baath et al. 1995; Johnson et al. 2010; Ojanen et al. 2017; Slesak 2013; Slesak et al. 2010; Ussiri and Johnson 2007). Land use, disturbance, and management do not always have negative impacts on SOC, however; SOC stocks can in some cases be increased via mechanisms that are the reverse of these negative impacts. For example, minimizing soil disturbance and erosion through less frequent tillage or the protection of the soil surface, promoting vegetation that sustains or increases organic matter inputs to the soil, and directly adding (or redistributing) surface organic matter are associated with sustained or increased SOC stocks in agricultural and forest soils (Guo and Gifford 2002; Vance 2000). Fires may result in an immediate net loss of SOC, but can also make the remaining SOC less decomposable, with the potential for longer-term feedbacks that promote SOC accumulation (Pellegrini et al. 2021).

In the Pacific Northwest, the relative importance of these mechanisms across land use and management regimes likely corresponds to the degree and duration of soil disturbance, with annual cultivation at one end of the spectrum, subtle biogeochemical shifts after forest harvesting at the other, and combinations of direct and indirect mechanisms for typical fires or harvests in the intermediate. That said, all of these mechanisms have considerable knowledge gaps, not least including why some appear to be more important in some settings than others. In this regard the mechanistic literature is much like the review literature on SOC management, in that both will benefit from analyses targeted at intermediate scales.

The importance and complexity of forests in the Pacific Northwest supports a commensurate complexity of scientific analysis, ecosystem management, and C management recommendations (Creutzberg et al. 2017; Hudiburg et al. 2013; Hurteau et al. 2019; Law et al. 2018). Against this backdrop, synthesis techniques such as meta-analysis have the potential to inform the discussion of C management in the Pacific Northwest. The present synthesis, representing the third in a series of ecoregional assessments, is intended to contribute to this discussion in the Pacific Northwest. It was motivated by four objectives. First, establish context for disturbance and management impacts on SOC stocks by assessing how SOC varies according to existing patterns of, e.g., climate and vegetation. Second, quantify the magnitude and variability of disturbance and management impacts on SOC stocks. Third, qualify these quantitative estimates using complementary approaches where possible, in order to assess confidence in them. Finally, provide scientifically defensible operational considerations for natural resource professionals wishing to incorporate SOC into their planning and management.

2. Methods

2.1 Study area- For the purposes of synthesizing data from the Pacific Northwest in an ecologically meaningful context, we defined the study area as all of the ecological sections present in OR and WA, and in some cases extending into adjacent states (Figure 1). Ecological Sections tier beneath the Province level in the U.S. Department of Agriculture-Forest Service (USDA-FS) ECOMAP hierarchical ecosystem classification system (Cleland et al. 1997; McNab et al. 2007). This definition of the study area includes a total of 17 sections, some of which extend into portions of CA, NV, and ID of the same topography and climate. Section descriptions are beyond the scope of this paper and are available in McNab et al. (2007).

2.2 Approach- In this analysis, we used synthesis methods described in detail in other recent papers (Nave et al. 2021; Ontl et al. 2019). These methods are four-fold: (1) effect size meta-analysis of data from published literature; (2) synthesis of soil pedon observations with remote sensing information; (3), analysis of national forest inventory (NFI) data from plots in which soils, biomass, and other ecosystem properties were measured; (4) literature review of strategies, approaches, and tactics of forest C management. Brief overviews of these methods follow below.

2.3 Meta-analysis- We synthesized data from 46 papers identified through literature review, which are summarized in Appendix S1: Table S1. We limited searches to 2008-2019 in order to add papers found through new searches to those already in our database from prior larger-scale meta-analyses (Nave et al. 2009; 2010; 2011; 2013). To be included, each paper had to: 1) report control and treatment values for SOC stocks or concentrations, 2) provide adequate metadata to constrain locations and use as potential predictor variables, 3) present response data not included in previous studies, and 4) be located within our Pacific Northwest study footprint (section 2.1). Twenty-two papers (of 1,880 reviewed) met these criteria, in addition to 24 pre-2008 publications from our database.

We extracted control and treatment SOC values from all 46 papers within the updated database and used these to calculate effect sizes (as the *ln*-transformed response ratio *R*). We used unweighted meta-analysis to estimate effect sizes and bootstrapped 95% confidence intervals (Hedges et al. 1999) using MetaWin software (Sinauer Associates, Sunderland MA, USA). We selected unweighted meta-analysis to maximize data availability (weighted meta-analyses require sample size and variance statistics in every paper), and because we did not assume that the data met the parametric preconditions of a weighted meta-analysis. Treatments of interest included fire management (wildfire and prescribed fire), silvicultural operations (harvesting, site preparation, or fuel management treatments), land use change (i.e., reforestation after cultivation or pasture), and soil amendments (additions of fertilizers, herbicides, or organic materials). More specific examples of these treatments of interest are provided in Appendix S1: Table S2, which details the attributes extracted from each published study.

We standardized response data using correction factors and prediction equations to convert: 1) samples analyzed using loss on ignition (LOI) as a metric of SOM; 2) SOC values reported as concentrations rather than the SOC stocks of interest to our analysis. Correction factors (%SOC=0.5*%LOI) and prediction equations (for estimating bulk density from C concentration) are described in Appendix S1.

We extracted predictor variables from each paper to identify factors that mediated treatment effects on SOC stocks. We looked up missing information (e.g., study site characteristics) in other publications from the same sites, or using information about the soil series reported from those study sites obtained from USDA-Natural Resources Conservation Service (USDA-NRCS) Official Soil Series Descriptions. Given the lack of standardization across studies in reporting details such as soil sampling depth and parent material, it was necessary to create categories for many attributes, in order to parse variation within and between studies into sufficiently replicated groups for meta-analysis. Our strategy for categorizing reporting depths requires specific attention here. First, we recorded the genetic horizon (e.g., Oe, Oa, A, Bw1) or sampling increment (as depth range in cm) for each SOC value. Next, for soils reported as depth increments, we correlated each specified depth increment to its probable genetic horizon, based upon associated methods descriptions or USDA-NRCS soil series descriptions. Last, we aggregated these into master horizon groups (i.e., O, A, B, BC and C, or mixtures of A/E/B horizons) for use as the categorical variable corresponding to soil depth. When SOC was reported for increments greater than 50 cm total depth, we summed them and categorized them as “whole profiles.” Average horizon thicknesses for these categorized master horizon groups corresponded with the other two data sources used in this analysis (Appendix S1: Table S3).

Similar to Nave et al. (2021), our efforts to obtain predictor variables and assign studies to groups were more involved than past analyses (e.g., Nave et al. 2010), but we used the information in essentially the same way. Namely, we used meta-analysis to identify significant predictors of variation in SOC responses, which is done statistically by parsing variation into within-group (Q_w) and between-group heterogeneity (Q_b), and inspecting corresponding P values. Grouping variables that have large Q_b relative to Q_w are significant ($P < 0.05$) and explain a larger share of total variation among all studies (Q_t). However, the statistical significance of P values is only one way to assess significance of meta-analysis results. In our meta-analysis, we were as interested in identifying groups that are significantly different from zero percent change (e.g., in response to harvest), in terms of their 95% confidence intervals, as we were interested in groups that were significantly different from each other (e.g., soil textures differing in their responses to harvest).

2.4 Synthesis of pedon and remote sensing data- We complemented the experimental strength of meta-analysis, which generates high-confidence inferences for a limited number of sites, with a synthesis of data for >1,700 locations across the study area. These data came from geo-located soil pedons from the USDA-NRCS National Cooperative Soil Survey (NCSS) Database, and included latitude, longitude, soil taxonomy, and physical and chemical properties of individual genetic horizons according to Schoeneberger et al. (2012) and Burt et al. (2004). Data from the NCSS Database span many decades of soil survey; to synthesize geo-located pedons with remote sensing information, we only used pedons from 1989-present so that pedons could be matched to independent, temporally discrete GIS products in the same manner as Nave et al. (2018; 2019; 2021).We extracted the following attributes for geo-located NRCS pedons, from spatial data products detailed in Appendix S1: Section S1.2, including land cover classification, vegetation type, aboveground biomass C stocks, historical fire data, mean annual temperature and precipitation (MAT and MAP, respectively), and topographic parameters including elevation, slope, aspect, and several derived topographic indices. Regarding land cover and burn data specifically, we used spatial data products as a starting point, manually inspecting aerial imagery for each sampling point to infer evident land use, past management, or fire, so that we could incorporate these activities as potential predictor variables in our analyses. Following these synthesis steps, our dataset for analysis included 1,722 pedons, spanning all land use, cover, and management conditions. Prior to beginning statistical analyses, which are focused on wildland ecosystems with woody vegetation (i.e., forests, woodlands, shrublands) we narrowed the dataset further, excluding pedons from developed, pasture/hay, grassland, cultivated, and barren land cover classes. The final dataset used in statistical analyses contained 1,146 pedons.

2.5 NFI dataset- We further complemented our meta-analysis and NRCS pedon + remote sensing datasets with an additional, independent observational dataset derived from the USDA-FS National Forest Inventory (NFI). The NFI plots that are the basis for data from the Forest Inventory and Analysis (FIA) program derive from an equal-probability sample of forests and woodlands across the CONUS. There is one permanent plot on approximately every 2,400 ha across the U.S., with each plot placed randomly within a systematic hexagonal grid (McRoberts et al. 2005). Soils are sampled from a subset of these plots, according to a protocol in which the forest floor is first removed, and mineral soils are then sampled as depth increments of 0-10 and 10-20 cm. The NFI plot design ensures that FIA data have no systematic bias with regard to location, ownership, composition, soil, physiographic or other factors. For this analysis, we queried the FIA Database for records of forest floor and mineral soil SOC stocks (Mg C ha^{-1}) for all single-condition plots in the ECOMAP ecological sections comprising the study area. We set the single-condition criterion in order to exclude plots divided along sharp boundaries into conditions of different stand age, slope, wetness, etc., such that local variation in such factors would misrepresent conditions at the actual location of soil sampling. 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 FIA data reasonably concurrent with the NRCS pedon and remote sensing data described above. Altogether, our datasets for forest floors and mineral soils were based on 194 and 130 NFI plots, respectively.

2.6 Statistical analysis of NRCS and FIA data- To complement the non-parametric meta-analysis of published literature data, we used data transformations and parametric statistics (SigmaPlot, SYSTAT Software, San Jose, CA US) to analyze NRCS and FIA data. These two observational datasets derived from fundamentally different sources, but they were sufficiently similar to be analyzed using a consistent set of techniques. Owing to their typically right-skewed distributions, we used *ln*-transformations as necessary to normalize response variables (though we report results as back-transformed means and 95% confidence intervals). We used ANOVAs with Fisher's Least Significant Difference to test for significant differences between *ln*-transformed group means, e.g., for SOC stocks under different vegetation types or management treatments. We used best subsets regressions to identify categorical (coded as dummy variables) and continuous variables (normalized and standardized by subtracting the mean and dividing by the standard deviation) explaining variation in SOC stocks in two reporting depths: A horizon vs. soil profile to 1m. We selected the optimal model for each depth as the one with the highest adjusted R^2, and comprised entirely of variables with significant partial P values. We set these criteria in order to identify the largest possible suite of factors influencing SOC stocks in each depth, while protecting against over-fitting by including variables that increased total proportion of variance explained, but themselves lacked significant relationships with SOC stocks. In all parametric statistical analyses, we set $P < 0.05$ as the *a priori* threshold for accepting test results as statistically significant.

3. Results

3.1 Ecoregional context

Across the Pacific Northwest, combined soil and aboveground biomass C stocks in ecosystems with woody vegetation (forests, woodlands, and shrublands) spanned a ten-fold range (Table 1; NRCS pedons + remote sensing information), from mean densities approaching 500 Mg C ha^{-1} in forests of the Oregon and Washington Coast Ranges down to <50 Mg C ha^{-1} in the shrublands of the Northwestern Basin and Range. More of this variation was driven by aboveground biomass C than by SOC stocks, which nonetheless ranged widely from 281 to 45 Mg C ha^{-1} (to a depth of 1 m) for these two ecological sections. In some ecoregions, ecosystems stored nearly twice as much C in aboveground biomass as in soil (e.g., Klamath Mountains, Puget Trough), while in others more of the C was held in soils than biomass (e.g., Oregon and Washington Coast Ranges, Southern Cascades).

Out of 14 potential predictor variables, variation in SOC stocks across the study area was a function of vegetation, climate, and topographic parameters (Table 2), whether considering the topsoil (A horizon) alone or the full profile to a depth of 1 m. Soil C stocks at both depths were most influenced by the same four parameters. These included categorical variables for two woodland/shrubland vegetation divisions (3.B.1.Ne Western North American Cool Semi-Desert Scrub & Grassland; 1.B.2.Nc Western North American Cool Temperate Woodland & Scrub), both of which were associated with significantly smaller SOC stocks than 1.B.2.Nd Vancouverian Cool Temperate Forest, which was the reference group for the categorical vegetation division parameter in the model. Soil C stocks increased significantly with mean annual precipitation, and were significantly less on south-facing slopes than on north-facing slopes (which served as the reference group for the categorical slope aspect parameter). Variation in SOC stocks was more predictable for A horizons than whole profiles (adjusted R^2=0.48 vs.

0.40), due to inclusion of three additional topographic and climatic variables in the strongest model meeting our selection criteria. Namely, east-facing slopes had significantly less A horizon SOC than the reference north-facing slopes, while elevation and mean annual temperature also showed negative correlations with SOC although these two variables appeared to be modestly autocorrelated (variance inflation factors of 3.44 and 3.08, respectively).

3.2 Disturbance and management effects on SOC stocks

Meta-analysis of relevant literature revealed that overall, fires significantly decreased SOC storage, while harvest had no significant overall impact on SOC stocks (Figure 2; treatment $P <$ 0.001). Significant C losses with fire (all types collectively) were detectable in all measured portions of the soil profile (Fig. 2A), from detrital O horizons, to topsoil A horizons and the deeper B horizons, despite their greater depth within the profile. In contrast to the negative impacts of fire, harvesting had no significant overall effect on SOC stocks in any horizon or at the whole profile level (Fig. 2B).

Analyses of complementary NRCS and FIA data validated meta-analysis results related to fire, but were ambiguous with regard to harvest. Across the two data sources, O horizon C stocks of control vs. harvested forests did not differ (Figs. 3A, 3B), but in the FIA dataset, burned O horizons held significantly less C than control O horizons. Mean O horizon C stocks among NRCS data, which did not differ between control and burned, are superficially misleading; O horizons were reported for only 24% of burned sampling locations, suggesting that fire may have eliminated many O horizons outright. In A and B horizons, the limited FIA data resolved no impacts of harvest or fire on SOC stocks (Figs. 3D, 3F), but the more abundant NRCS observations revealed significantly lower SOC stocks in burned A and B horizons than in controls (Figs. 3C, 3E). The more regionally abundant NRCS data also resolved significantly larger SOC stocks in the A horizons of harvested forests than unharvested controls. However, underlying data structure indicated two vegetation divisions showing contrasting responses to harvest, with significant harvest-associated gains in 1.B.2.Nd Vancouverian Cool Temperate Forest masking significant losses in 1.B.2.Nb Rocky Mountain Cool Temperate Forest (Appendix S1: Fig. S1).

Analyses of NRCS and FIA data also allowed insight into several frequently encountered disturbance and management conditions not tested using meta-analysis (Figure 3). Namely, insect and/or disease-damaged stands showed no SOC differences from controls in FIA data in any reported soil layer (Figs. 3B, 3D, 3F). Stands exhibiting physical damage from livestock activity had significantly less O horizon C than controls among FIA observations. More intensive agricultural use (past cultivation) was associated with smaller SOC stocks in all horizons among NRCS pedons (Figs. 3A, 3C, 3E), although this difference was not statistically testable for O horizons due to the rarity of O horizons in forests growing on previously cultivated lands.

3.3 Factors influencing disturbance and management impacts on SOC stocks

Meta-analysis indicated little in the way of explainable variation among harvesting studies. Harvest-related variables (e.g., intensity of cutting, amount of material removed, time since harvest) and site factors (e.g., parent material, ecological section) were not statistically significant predictors of variation in harvest effects. In the case of several predictor variables, an individual group (e.g., a single parent material type) was significantly different from 0% change even though the corresponding predictor variable was not statistically significant, but such results were associated with small sample sizes. Indeed, only one factor showed a statistically significant overall influence on SOC responses to harvest; namely, meta-regression indicated that harvest impacts on O and A horizons (considered collectively) were positively related to mean annual temperature. This effect (*P* = 0.015, explaining 7% of variation between response ratios), suggested an approximate temperature threshold, with harvesting above ~8.5° MAT associated with increasingly large SOC gains, and harvests falling farther below the temperature threshold showing increasingly large SOC losses (Appendix S1: Fig. S2). NRCS O and A horizons (also considered collectively) from harvested vs. control groups showed a similar pattern (Appendix S1: Fig. S2), with SOC stocks increasing with temperature (regression $P < 0.001$), and harvested observations showing significantly steeper, more positive slopes than control observations ($P < 0.001$).

In contrast to harvest studies, meta-analysis revealed several significant sources of variation underlying the overall significant effect of fires. Across all observations, wildfires produced significantly larger (and more variable) negative impacts than prescribed fires (Figure 4A; *P* = 0.046), a pattern that was mirrored in comparisons of high-severity to low-severity burn areas regardless of fire type (Figure 4B; *P* = 0.013). In terms of the depth of the negative impacts, wildfires produced significant SOC losses in all parts of the soil profile (Figure 4C), while negative impacts of prescribed fire were restricted to O horizons only, where they were modest and less variable. In wildfires, steep slopes had significantly larger O horizon losses than level to gentle slopes (*P* = 0.001).

Several additional soil, fuel, and silvicultural treatments were also testable using meta-analysis of published literature, which revealed a range of SOC impacts from neutral to positive (Figure 5), though without sufficient replication to test for the depth distribution of effects. Among treatments implemented specifically to reduce fuels, harvesting (any intensity, with or without residue removals) followed by residue or site preparation burning was associated with no significant change in SOC storage, while a fell-leave-burn treatment (from a juniper control study) was associated with a modest but statistically significant increase in SOC (Fig. 5A). In terms of non-extractive practices, the establishment of N-fixing vegetation (specifically red alder), addition of surface organic amendments (e.g., biochar, wood chips, biosolids) or inorganic N fertilizers all significantly increased SOC stocks (Fig. 5B). The use of herbicides for competing vegetation control, and additions of other inorganic fertilizers or urea did not have significant effects on SOC storage.

4. Discussion

4.1 Inferences and implications

Using multiple approaches creates more robust inferences into management and disturbance impacts on SOC, but also introduces complexity into their discussion. We manage this complexity by organizing our discussion around Table 3, which summarizes the key findings of our synthesis and subjectively rates our confidence in each one, based upon the consistency or degree of support across datasets and approaches. Our most important finding is that variation in SOC stocks across the region is due to ecological factors such as topography, climate, and vegetation type; land use, management, and disturbance terms were not even close to being included in optimal models of SOC storage. However, close correlations between ecologic factors and baseline SOC stocks do not mean these relationships are static. In an era of increasing climate change and fire, the topographic, climatic and vegetation patterns in SOC that we report (less SOC on shrublands and S-facing slopes, declines with warming or decreasing precipitation) suggest that SOC is vulnerable to continued intensification of climate change and wildfire (Littell et al. 2009; Schoennagel et al. 2017). Furthermore, the direct and indirect impacts of climate change, fire, drought, heat, topographic exposure, and vegetation dynamics on SOC are not limited to the surface; they are significant at the whole profile level. The depth of this potential problem and related impacts on ecosystem services thus requires that SOC is included in the discussion of climate change, disturbance, and forest C management in the Pacific Northwest (Abatzoglou et al. 2016; Halofsky et al. 2020; Meigs et al. 2009; Raymond et al. 2015).

The three approaches we used to address fire impacts on SOC stocks converged on similar general findings, with meta-analysis of published studies producing the most detailed results. Meta-analysis is a strong technique for addressing a highly variable ecological process like wildland fire because it can: 1) quantify trends too variable for detection within individual studies by synthesizing across them, and 2) use variability within and between studies to probe sources of variation in the overall effect. In terms of general findings, all three approaches indicated that wildland fires significantly decreased SOC storage. Where the two observational datasets resolved significant effects of fire, effect sizes were close to the estimates of average SOC stock changes derived from meta-analysis: O horizon, -66% for FIA data and -64% for meta-analysis; A horizon, -8% for NRCS data and -12% for meta-analysis; B horizon, -7% for NRCS data and -14% for meta-analysis. Due to limited replication, observational data could not be used to address variation in fire effects; in contrast, meta-analysis indicated that wildfires (and severe burn locations, regardless of fire type) showed significantly larger and more variable negative impacts than prescribed fires and low-severity burn areas, respectively. These results reflect the complex interactions between fuels, topography, and meteorological conditions that drive fire behavior (Finney et al. 2011; Sullivan et al. 2017), and suggest that in the case of SOC impacts, variability in losses may be as important as their magnitude. In a wildfire, severely burned areas suffering the most extreme SOC losses may represent a small percentage of the area but a disproportionate share of the overall C losses associated with the fire, and present the most pernicious threats to ecosystem services and forest re-establishment (Burke et al. 2021; Seidl et al. 2016). Moreover, the significant loss of SOC in all parts of the profile with wildfire, vs. the loss of only O horizon C with prescribed fires, is another indication of the potential for wildfires to have farther-reaching, longer-term impacts on soil productivity, ecosystem services, and forest C. Across the range of soil depths impacted by fire, the time required to accumulate existing pools of C varies by orders of magnitude; O horizons hold C that has typically accumulated over years to decades, while much of the SOC held in A and B horizons has accumulated over many centuries or even millennia (Crow et al. 2007; Heckman et al. 2013). Although our statistical approaches did not detect any timeframe of SOC recovery after fires, these estimates of C residence times suggest the potential for very long replacement times of lost SOC, and accordingly, long-term net reductions in total forest C with fires.

Published studies from the Pacific Northwest provide nuance that complements our overall findings and offer deeper insights indicating that the impacts of wildfire may be more severe than can be detected using typical methods. The strongest example of this emerges from the 2002 Biscuit Fire in the Klamath Mountains of southwestern Oregon. This extreme, well-studied event burned through an existing experimental footprint, allowing for precise quantification of pre- vs. postfire SOC stocks. In the first published study from the event, Bormann et al. (2008) documented significant losses of soil volume, including mineral soil material, some of which was likely lost in the fire itself through entrainment into the smoke plume. Had pre-fire data not been available, the estimated loss of SOC would have been underestimated by half, due to failing to account for the losses of soil material that are not detectable with typical fixed-depth sampling designs. For our meta-analysis, which cannot address the specific mechanisms of SOC loss, we obtained SOC data for control, burned (low- vs. high-severity; Heckman et al. 2013), and harvested (lightly vs. heavily cut; some burned, some not; Homann et al. 2015) stands in the landscape affected by the Biscuit Fire. These studies offer detailed insights into factors that contributed to this historic fire and its consequences. In terms of consequences, Heckman et al. (2013) used highly sensitive radiocarbon analysis to demonstrate that the forest floor loss and disappearance of 2 cm of topsoil estimated by Bormann et al. (2008) was also associated with a change in the nature of the SOC; the SOC that (partially) replaced the stable forms of SOC lost during the fire consisted of more recent, rapidly cycling substrates. Furthermore, soil radiocarbon and charcoal data suggested that areas burned at higher severity during the Biscuit Fire may have historically burned more severely or frequently than areas burned at low severity – a potential positive feedback which was to some degree related to past management. Areas that burned in a 1987 fire or were salvage harvested and replanted thereafter burned more severely in the Biscuit Fire (Thompson et al. 2007). Furthermore, Homann et al. (2015) showed that while harvest alone did not affect SOC on this landscape, fire-induced losses in SOC due to the Biscuit Fire were significantly larger in stands that were more heavily harvested years before. Collectively, these patterns from one set of complementary landscape-level studies indicate that specific management decisions can impact the probability, severity, and SOC consequences of later wildfires. These patterns are consistent with our regional findings but also reveal the importance of considering management opportunities and constraints within the context of disturbance histories, ecosystem trajectories, and soil characteristics of landscapes (Zald and Dunn 2018). Compound disturbances, especially when they include extreme wildfires like the Biscuit Fire, can have markedly greater impacts on SOC than expected based upon single-disturbance studies. Fires as severe as the Biscuit Fire are not representative of wildfires in general, given that >95% of wildland fires are extinguished while small (NIFC 2021), before growing into the large, severe events that are subsequently and disproportionately the focus of research. Nonetheless, as climate change and fire severity continue to increase across the region (Halofsky et al. 2020), compound disturbances and related positive feedbacks to SOC are likely to continue to increase.

In contrast to fire, the approaches we used revealed few statistically significant effects of harvest or fuel reduction treatments on SOC stocks, and little in the way of explainable variation. Exceptions to the general lack of effects were a modest meta-analytic increase in SOC after a fell-leave-burn juniper control treatment from a single publication, and an indication of an overall, regional-average SOC increase with harvesting (compared to control) in A horizons for one of the two observational datasets (NRCS). On the other hand, published papers and NRCS pedons both suggested that SOC tended to decrease with harvesting in colder climates, increase in warmer locations, and remain unchanged near the middle. This could reflect confounded relationships between temperature and elevation, which were noted through elevated variance inflation factors in the A horizon linear model (Table 2). The temperature dependence of harvest impacts may also be related to the apparent vegetation dependence noted in the NRCS dataset. The Vancouverian Cool Temperate Forest division, where harvest was associated with increased A horizon SOC, has significantly warmer MAT than the Rocky Mountain Cool Temperate Forest division (9.2° vs. 6.8°), where harvesting was associated with statistically significant A horizon SOC declines. Unfortunately, because these vegetation divisions differ in many management and ecological factors (including climate), the mechanism (s) for their divergent harvest responses cannot be disentangled. Regardless the explanation(s) for this pattern, judicious, place-based management will mitigate the potential for harvest losses of SOC in the colder, drier interior portions of the study area. More broadly at the regional scale, the ambiguous evidence for directionally variable, modest magnitude harvest impacts stand in stark contrast to the consistently deep, large, and negative impact of wildfires.

The implications of our analysis carry the most weight on an ecoregional basis, where the increasing number, frequency, extent, and severity of wildfires (Littell et al. 2009; Schoennagel et al. 2017) suggest an intensifying, widespread impact on forest SOC as wildfires continue to increase. Our synthesis provides a clear distinction between management approaches from a C accounting perspective: in the context of widespread elevated fuel loads and chronically lagging reforestation efforts following fire (Cook-Patton et al. 2020; Domke et al. 2020; Dumroese et al. 2019; Franklin and Johnson 2012; Haugo et al. 2015), combinations of mechanical thinning and prescribed fire may be the most effective means to jointly minimize fire risk and SOC loss, especially in dry interior forests (Halofsky et al. 2020). Careful vegetation management can remove and utilize C from forests in a controlled fashion (Dugan et al. 2018; Fain et al. 2018; Malmsheimer et al. 2011) while protecting against uncontrolled losses of C from soils and biomass, even as rapid reforestation creates opportunities for C recovery and gain (Nave et al. 2018; 2019c). These findings will ideally inform strategic, regional discussions of C stewardship in forests of the Pacific Northwest. At the same time, site-level operations—whether in fire suppression, fuel treatments, or forestry—are implemented according to project- or event-specific constraints that cannot be addressed in the necessary site-level detail using synthesis techniques.

4.2 Limitations

The inferences of our study are limited in two important ways. Regarding the first limitation, our inferences into the timescales and temporal dynamics of SOC change are limited by the existing data and our ways of using them. Published papers provide the best opportunity to constrain temporal patterns, because they usually report time since disturbance. However, this information is not always known or reported, and compound disturbances defy a singular, simplified definition of time since disturbance. Constraining time since disturbance is even more challenging for our two observational data sources (NRCS pedons, FIA plots). With few exceptions, these sources lack observations of the timing of harvest, fire or other disturbances, whether in recent decades or before remote sensing and plot inventories began.

Problematic study designs introduce a second limitation into our analysis. Our timescale of interest extends into the centuries over which most SOC cycles, yet most published studies are conducted within 5 years of a harvest or fire, and long-term studies typically reach no further than 20-30 years. Those studies that do make comparisons of SOC across multi-decadal or multi-century timescales rely on indirect observational designs, such as chronosequences, which carry well-known potential pitfalls of interpretation (Yanai et al. 2000). Quite often, the rarity of old, unmanaged reference forests means that they differ from younger managed forests in more ways than just disturbance history (e.g., elevation, snowfall, mean annual temperature, and soil order in Law et al. [2001]). It is therefore often difficult to know whether the best available reference actually represents the potential for long-term SOC recovery after disturbance, vs. the equilibration of SOC stocks to inherent differences in ecological factors. Observational data sources (NRCS, FIA) are plagued by the same problem, though primarily with regard to the disturbed condition: the great majority of FIA plots and forested NRCS pedons show no obvious evidence of recent harvest or fire. Plots or pedons that have been recently harvested or burned are one to two orders of magnitude less common. In our statistical comparisons of these data, we assume that control (i.e., not recently disturbed) and treatment (i.e., harvested or burned) observations are distributed randomly with regard to the ecological factors that influence SOC stocks. If this assumption is not valid, then the unbalanced data distribution may cause the actual influence of management or disturbance to deviate from our estimates.

These two inferential limitations—short-term timescales and problematic study designs—may also limit the implications and applications of our synthesis. Problematic studies that confound ecological differences with disturbance or management histories increase uncertainty in our estimated disturbance or management effect sizes, even if they do not directionally bias them. The overall ecoregional effect sizes of fire across our three data sources (section 4.1) provide an example. Meta-analysis of published papers appears to overestimate the effects of fire on mineral soils (relative to NRCS pedons), which may indicate that published papers tend to be biased towards larger, more severe fires. If this is the case, then the SOC implications of fire may be overemphasized. On the other hand, considering organic and mineral soil horizons collectively, there is remarkable agreement between meta-analysis and NRCS pedons: directionality of SOC change is consistently negative, and estimated effect sizes differ by 2-7% across a range extending from 7-66%. In this regard the primary limitation is less one of bias than of uncertainty; even a 5% underestimation of fire-induced SOC losses can translate to Tg of C at an ecoregional scale. In terms of timescale, the short-term nature of the available data challenges the application of our results to forest management. Using fire as an example once again, our key finding that wildfires decrease SOC throughout the profile (Table 3) is derived mostly from studies <10 years since fire. This short-term focus may overestimate impacts of fire on SOC by over-representing immediate responses, especially in surface soils that are most directly impacted (e.g., O horizons). Furthermore, the lack of long-term data (<25% of studies address timescales beyond 25 years) means that we are unable to quantify long-term rates of SOC recovery. Thus, the management applications of this high-confidence, short-term finding are questionable. Do low vs. high-severity wildfires have the same recovery times of lost SOC? Are some forest types or topographic settings faster to recover? Within the limitations of current data availability, these questions cannot be answered with confidence in ways that are both systematic and localized. In order to address these questions, their management applications, and tactical considerations, it is necessary to combine our synthetic approach with further literature review.

4.3 Management applications

As in other ecoregions (Nave et al. 2019b; 2021), place has a stronger influence than practice on SOC stocks in the Pacific Northwest, where SOC stocks span an order of magnitude across ecological sections (Table 1). Within these ecological sections and on down to landscape levels, topography, climate, and vegetation are the key drivers of variation in SOC (Table 2). Biomass C stocks respond to the same drivers, grossly scale with SOC stocks, and constrain management and disturbance regimes across the ecoregion. At the highest level, the region bifurcates into wet vs. dry systems approximated by the Vancouverian vs. Rocky Mountain Cool Temperate Forest vegetation divisions that showed some evidence of divergent SOC responses to harvest (Appendix S1: Fig. S1). The wet, productive Vancouverian systems, which are prevalent in the Coast Ranges and Western Cascades, have large C stocks (Table 1), low frequency, high severity fire regimes, and are primarily managed for Douglas-fir and other mesic conifers. Here, management has resulted in landscapes where relatively even-aged, mid-seral stands occur in excess of the natural range of variation (Donato et al. 2020). Given that wildfire negatively impacts SOC, while harvesting has neutral to potentially positive impacts, strategic, landscape-level harvesting to reduce stand connectivity and structural homogeneity may be a way to protect against severe fires and resulting SOC losses in the long term.

Drier interior forests have also deviated from their historic disturbance regimes, ecosystem and landscape structures, but have smaller SOC and biomass C stocks (Table 1; DeMeo et al. 2018; Hessburg and Agee 2003). Here, there have long been calls for active management to restore forests to their natural range of variation (Mutch et al. 1993; Peterson and Halofsky 2018), yet these systems have some potential to lose SOC with harvesting. The basis for potential losses is uncertain, but may include climatic limitation of primary production (smaller C inputs to soil; slower regrowth after harvest) or differences in soil properties. Dry forests throughout the interior Pacific Northwest grow on soils with a volcanic ash cap, which due to their unique texture and structure makes them susceptible to physical disturbance during mechanized operations (McDaniel and Wilson 2007). Structural damage and physical disturbance are not the only potential drivers of SOC loss following harvest—biogeochemical shifts likely alter SOC cycling too—but when and where they happen, detrimental physical impacts (e.g., erosion or forest floor displacement) probably cause SOC losses. Thus, in dry interior forests, site-specific tactical adjustments to harvesting or fuel reduction operations may play a more prominent role than in the wetter, west-side forests where management may be more strategically aimed at mitigating severe fire probability.

Management options for mitigating wildland fire potential, enhancing forest climate adaptive capacity, and protecting soils from disturbance during operations have been well articulated in existing publications that are focused on or relevant to the Pacific Northwest (Angima and Terry 2011; Crawford et al. 2021; Nash et al. 2020; Peterson and Halofksy 2018). These summaries provide well-supported actions for minimizing some of the potential drivers of SOC loss.

However, none of these references explicitly address C. The *Practitioner's Menu of Adaptation Strategies and Approaches for Forest Carbon Management* (Ontl et al. 2020) explicitly addresses C, though primarily aboveground and at a high (strategic) level. The *Menu* provides a framework for incorporating SOC explicitly, at a tactical level, into management discussions in the Pacific Northwest. In Appendix S1 (Appendix S1: Table S4), we offer a set of example SOC management tactics, based on the results of our analysis, as supported by additional relevant literature. Recognizing that potential tactics are practically limitless, we provide a focused, defensible subset, the majority of which fall under adaptation strategies involving reducing C losses from and enhancing forest recovery after disturbances.

Based on the findings of our analysis, fire is the principal disturbance of concern to SOC management in the Pacific Northwest, and the first two example tactics therefore address fuel and fire management. Example tactics related to fire recognize that in the wetter west-side forests with naturally low-frequency, high-severity fire regimes, fires are likely unavoidable; therefore, SOC management tactics align more with where to prioritize suppression or post-event recovery. In contrast, SOC impacts of fire in drier east-side forests with historically shorter fire return intervals may be proactively mitigated through fuel management intended to decrease fire probability or severity (DeMeo et al. 2018; Donato et al. 2020; Halofsky et al. 2018). In the case of these examples, the tactics themselves are quite different (reactive vs. proactive), even though they are based on the same underlying result of our analysis, which showed that steeper slopes are more likely to lose more O horizon C in wildfires (section 3.3).

Where proactive fuel management is conducted, our results (and supporting literature) highlight the potential role of prescribed fire in mitigating the probability of fire and SOC loss. There are numerous tactical adjustments that can be made to prescribed burning, especially in terms of location and timing. Hatten et al. (2008) compared prescribed fires conducted in spring vs. fall in the Blue Mountains, determining that while fall fires significantly diminished O horizon and mineral soil C stocks, spring burning had little to no detectable impact on SOC stocks. In that study, seasonality of fire impacts was attributed to the lower severity of fire during spring (relatively damp) vs. fall (relatively dry) fuel and soil conditions. Given that even spring burning effectively reduced fuel loads, the study indicated how careful burn prescriptions can support fuel reduction objectives while protecting against soil and other ecosystem impacts, which may be particularly important in dry, fire-prone, interior forests. Recent research in other interior dry forests with ash cap soils extends this finding, showing that while thinning alone does not affect O horizon C stocks, prescribed fire or thinning + prescribed fire (with burning completed in late spring) decrease O horizon C stocks (Busse and Gerrard 2020). Importantly, that study also detected no negative impacts on other soil or ecosystem properties, including erosion potential and site nutrient capital. Both studies revealed that prescribed fires needed to be conducted on a roughly similar return interval as the historic fire regime (<20 years) to maintain down woody and O horizon fuel loads. Because the extent of overstocked, fuel-dense forests is large and this return interval is short, it may be necessary to prioritize such prescribed burns. Our fire results suggest that priority may lie in the most steeply sloping landscapes, all the more since mechanical fuel removals are at greater risk of causing soil disturbance in such settings.

SOC management tactics relevant to harvesting and fuel reduction may be most critical in dry, east-side forests with their structurally sensitive ash-cap soils, smaller baseline SOC stocks, and potential for SOC losses. To the degree that physical disturbance drives SOC losses with harvesting, existing best management practices (BMPs) such as restricting operations on physically vulnerable soils (e.g., ash-cap or clay soils, slopes greater than 20%) or limiting traffic to certain times (e.g., frozen ground, low soil moisture) are sound tactics (Angima and Terry 2011; Crawford et al. 2021; Nash et al. 2020; Page-Dumroese et al. 2010). However, this is not to say that these tactics are irrelevant outside the dry interior. Managing soil disturbance maintains hydrologic function and stand production, which are especially important in the wetter coastal forests. The potential for compound disturbance impacts in wetter coastal forests, such as past harvesting increasing SOC losses in the Biscuit Fire (section 4.1), also illustrates why judicious adherence to BMPs may carry long-term SOC benefits even in forests with little risk of direct harvest-induced SOC loss.

On any given landscape in the Pacific Northwest, topographic position and exposure are key considerations in harvesting, fuel reduction, and SOC management. North-facing aspects, which are climatically more mesic than other exposures, often support higher productivity, fuel density, and fire severity, especially at middle and upper slope positions (Birch et al. 2015; Dillon et al. 2011). Our analysis of NRCS pedon data likewise detected larger SOC stocks on north- than south- or east-facing slopes (Table 2), and data included in our meta-analysis revealed similar patterns. Griffiths and Swanson (2001) compared SOM contents of old-growth and harvested (5, 15, and 40 year old) Douglas-fir stands in the Western Cascades, noting no differences related to stand age, but significantly more SOM on north-facing slopes and at higher elevations compared to other topographic positions. These topographic patterns may justify heavier thinning or fuel removal on ridge tops and adjacent south-facing slopes, where SOC stocks are smaller to begin with and regeneration failure more likely in a warming, drying climate.

Not all of the example SOC management tactics in Appendix S1: Table S4 necessarily involve harvest removals or fuel management. As another example, red alder is associated with significantly larger SOC stocks in our meta-analysis of published literature, suggesting that it may stand out as a management opportunity for SOC gain. On one hand, red alder requires decades to have an effect on SOC, isn't feasible beyond mesic to hydric soil and climate conditions, and can have negative impacts on desired future stand conditions or production goals in managed forests, especially on high-fertility soils. On the other hand, red alder can enhance production of desirable species in strongly N-limited soils, and does not carry the fossil fuel C externalities of fertilizer (typically urea) production and application. On balance, promoting red alder over co-occurring hardwoods on sites that are not managed for commercially valuable conifers, or favoring its persistence at low densities on poor conifer sites may be compatible with multiple goals, including SOC sequestration at landscape levels. Overall, these and other tactics in our menu provide a defensible starting point for those intending to consider SOC in the context of disturbance, management, and forest C in the Pacific Northwest. As it is applied across range of conditions, the menu can be augmented, refined, and modified in response to new research, monitoring, mapping, and other decision support tools.

Acknowledgments

We thank Alexander O'Neill and Nickolas Viau for their careful literature review and GIS work, respectively, Stephanie Connolly and Evan Kane for helpful discussions of soil organic matter and forest management, and two anonymous reviewers for feedback that improved this manuscript through peer review. This work was supported by the USDA-Forest Service, Northern Research Station, under agreements 17-CR-11242306-028 and 19-CR-11242306-096. We are also grateful to the Frank E. and Seba B. Payne Foundation and the University of Michigan Biological Station for support.

Open Research Statement

Datasets (Nave 2021) used in this study, which were derived from published papers and publicly available databases, are available from the University of Michigan Research and Data Hub: https://mfield.umich.edu/dataset/land-use-and-management-effects-soil-carbon-northern-us.

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Table 1. Ecoregional variation in C density (Mg ha^{-1}) in aboveground woody biomass, soils to 1m, and their sum and ratio for the forests, woodlands, and shrublands of each ecological section. Values are means, sample sizes, and 95% confidence intervals. Soil data are from NRCS pedons; biomass data were extracted from remote sensing information for pedon geolocations.

	AG. Biomass			**Soil Profile to 1m**				
Ecological Section	**Mean**	**n**	**95% CI**	**Mean**	**n**	**95% CI**	**Sum**	**AG:SOC**
OR & WA Coast Ranges	217	80	187, 247	264	156	243, 286	481	0.8
Western Cascades	246	31	207, 285	162	136	137, 193	408	1.5
Klamath Mountains	221	15	174, 269	117	41	94, 146	338	1.9
Northern Cascades	194	21	145, 242	134	51	104, 173	328	1.4
Puget Trough	220	22	160, 279	104	32	72, 148	323	2.1
Southern Cascades	113	22	75, 151	155	35	117, 205	268	0.7
Willamette Valley	97	62	75, 120	154	102	140, 170	252	0.6
Okanogan Highland	93	49	78, 108	94	111	86, 102	187	1.0
Blue Mountains	68	54	57, 79	114	105	103, 127	182	0.6
Palouse Prairie	61	5	1, 121	99	35	85, 116	161	0.6
Eastern Cascades	80	28	64, 95	80	69	65, 99	160	1.0
Modoc Plateau	47	10	15, 79	57	13	41, 80	105	0.8
Snake R. Basalts & Basins	12	7	0, 37	83	54	70, 99	95	0.1
Columbia Basin	19	4	6, 32	59	44	49, 72	78	0.3
Owyhee Uplands	20	3	0, 58	54	16	30, 97	73	0.4
Blue Mountain Foothills	10	41	3, 18	53	87	45, 62	63	0.2
Northwestern Basin & Range	0	34	0, 0	45	57	35, 58	45	0.0

Table 2. Sources of variation in SOC stocks of A horizons and whole soil profiles to 1m, based on best subsets regression analysis of NRCS pedons and co-located remote sensing attributes. The number of observations and adjusted R^2 are reported for each model. Parameters in each model are presented in descending order of the absolute values of their standardized coefficients, in order to visualize their relative influence on SOC stocks. The coefficient, standard error, *P* value, and variance inflation factor is presented for each parameter.

Variable	**Coef.**	**SE**	**P**	**VIF**
A horizons[a]				
Constant	3.997	0.0456	<0.001	0
3.B.1.Ne Shrubland	-1.107	0.103	<0.001	1.372
1.B.2.Nc Woodl./shrubl.	-0.514	0.181	0.005	1.07
MAP	0.38	0.0377	<0.001	1.398
South-facing slope	-0.33	0.0724	<0.001	1.145
East-facing slope	-0.246	0.0808	0.002	1.186
Elevation	-0.183	0.0502	<0.001	3.44
MAT	-0.124	0.0555	0.025	3.08
Profile to 1 m[b]				
Constant	5.007	0.0414	<0.001	0
3.B.1.Ne Shrubland	-0.652	0.1	<0.001	1.21
1.B.2.Nc Woodl./shrubl.	-0.553	0.182	0.003	1.06
MAP	0.444	0.0374	<0.001	1.24
South-facing slope	-0.274	0.068	<0.001	1.007

[a] n=1381, R^2=0.48, *Cp*=4.01.

[b] n=1146, R^2=0.40, *Cp*=-2.52.

Table 3. Synthesis summary. Major inferences have more (+) or less (-) confidence based on support across datasets; low-confidence or highly specific inferences are omitted.

Major Inference	Confidence	Management, C accounting, and policy considerations
1. Place influences SOC storage more than practice	+	Land use and management have a secondary influence on SOC stocks, which vary according to inherent ecological factors; carbon-informed planning and operations take into account these factors
2. Fires consistently decrease SOC storage	+	In general, fires decrease SOC stocks; the magnitude and variability of losses increase with fire severity
3. Wildfire decreases SOC stocks at all depths	+	Wildfire is associated with significant SOC declines in O, A, and B horizons, with potential negative feedbacks to soil productivity, vegetation recovery, and ecosystem services
4. Prescribed fire decreases C stocks only in O horizons	+	Prescribed fires have no impact on mineral soils (A or B horizons), and may thus decrease surface fuels with less impact on soil productivity than the impacts associated with wildfires
5. Fire-driven SOC losses may be exacerbated on warm, dry sites	-	Sites with S-facing exposure, less precipitation, and shrub-dominated vegetation store less SOC, may be more vulnerable to SOC loss directly with fire and through feedbacks or state changes
6. Fuel reduction treatments have minimal direct impacts on SOC stocks	-	Harvests with residue burning or fell-and-burn treatments have variable, but overall neutral or slightly positive effects on SOC; C policy & management considerations may change with time
7. Harvesting has limited impacts on SOC storage	-	Regionally, harvesting does not affect soil C; cooler interior dry forests may lose O&A horizon SOC while warmer wetter coastal lowland forests respond with gains
8. Red alder increases SOC stocks in managed conifer stands	+	Deliberately establishing alder, or allowing it to persist in managed stands or landscape mosaics has strongly positive SOC impacts; edaphic constraints and competing management objectives limit the regional suitability of this strategy
9. Organic amendments increase SOC stocks	+	Direct additions of biochar, biosolids, or chipped wood increase SOC stocks, and may have additional soil productivity benefits when used for restoration and rehabilitation of burned sites
10. Other amendments have minimal impacts on SOC stocks	-	Fertilizers and herbicides have variable but largely neutral to modest direct effects on SOC stocks; their effects on ecosystem C stocks may differ
11. SOC is diminished in forests with agricultural uses or legacies	-	Livestock grazing and cultivation have direct, negative impacts on SOC; these may exacerbate impacts of other perturbations on soil and ecosystem C stocks in forests with a history of agricultural use

Figure Captions

Figure 1. Map of study area. Shaded polygons are ecological sections. Numbered point locations (approximate) represent papers reviewed for the meta-analysis. The two smaller point sizes are papers with ecosystem-specific and landscape-level designs, respectively; the larger point size represents papers with sites spanning a subregional scale (see Appendix S1: Table S1). Red squares and blue triangles represent forest and woodland NRCS pedons and FIA plots (approximate), respectively.

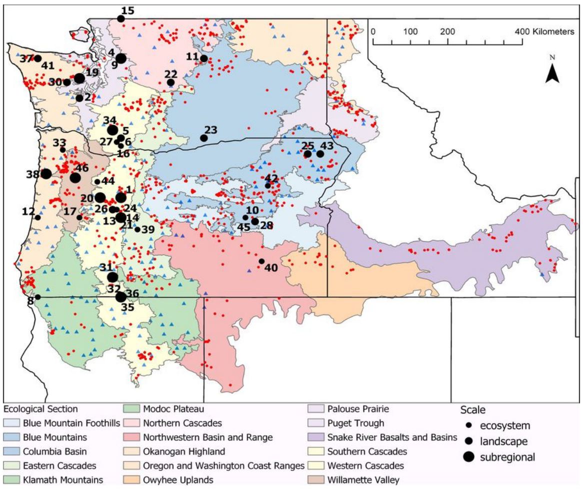
Figure 2. Proportional changes in SOC storage, by sampling depth, associated with fire (panel A) or harvesting (panel B), as quantified using meta-analysis of published literature data. Plotted are means, bootstrapped 95% CIs, and sample sizes; groups with CIs overlapping the dotted reference line show no significant change in soil C storage. Note that no published fire studies reported to a sufficient cumulative depth (50 cm) to provide data for the whole soil profile.

Figure 3. Soil C stocks in O (upper row), A (middle row), and B (bottom row) horizons, by land condition, for the observational NRCS (left column) and FIA (right column) datasets. Points plotted are means, 95% confidence intervals, and sample sizes. Letters denote significant differences between groups within each panel. Note: x-axes differ between horizons.

Figure 4. Proportional changes in SOC storage due to fire, as quantified using meta-analysis of published literature data. In panel A (left, above dashed line), points differentiate prescribed fire vs. wildfire; in panel B (left, below dashed line), points differentiate low-severity vs. high-

severity burn areas (as reported by paper authors) within fires of either type. In panel C (right), points indicate distinct parts of the soil profile, for prescribed fire (open circles) vs. wildfire (filled circles). Plotted are means, bootstrapped 95% CIs, and sample sizes; groups with CIs overlapping the dotted reference line show no significant change in SOC storage relative to controls.

Figure 5. Proportional changes in SOC storage with fuel management and other silvicultural practices, as quantified using meta-analysis of published literature data. Points in panel A (above dashed line) represent harvest and burn vs. fell-leave-burn treatments; points in panel B (below dashed line) represent non-extractive silvicultural practices. Plotted are means, bootstrapped 95% CIs, and sample sizes; groups with CIs overlapping the dotted reference line show no significant change in soil C storage.



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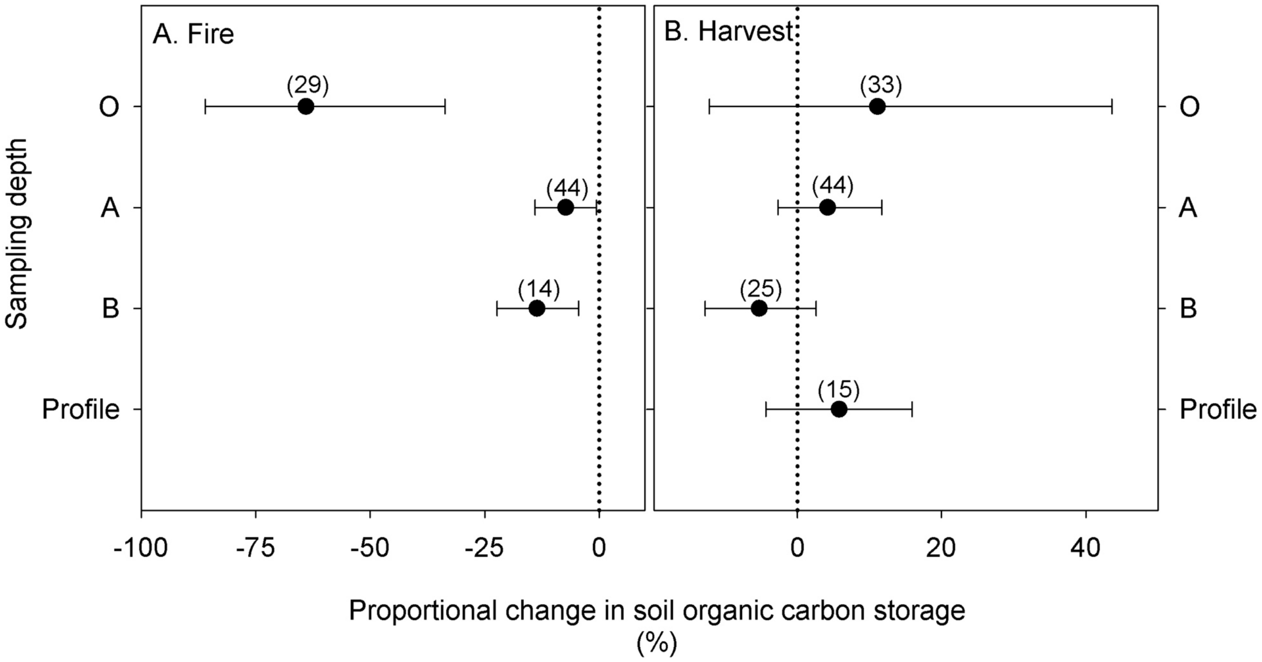

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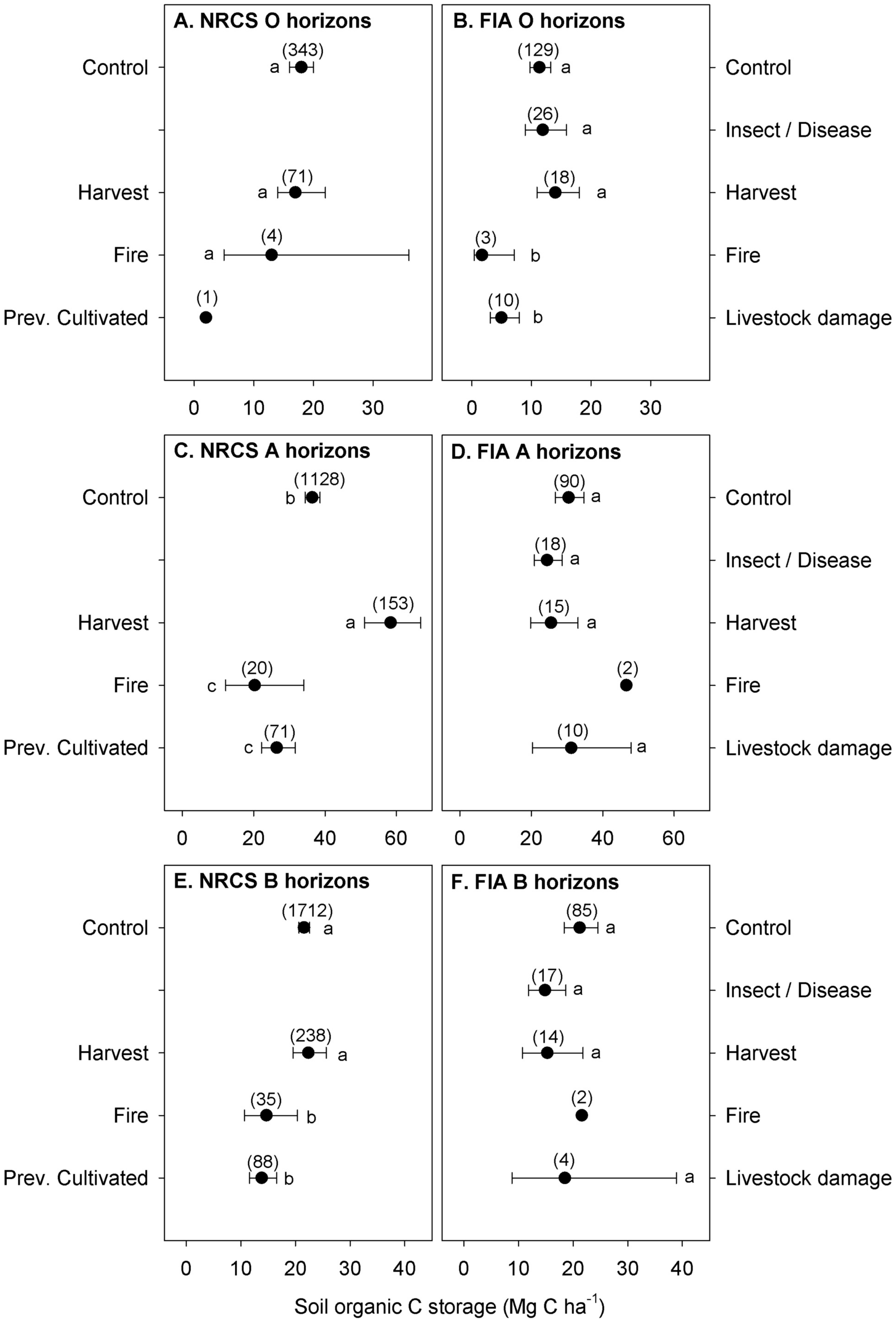

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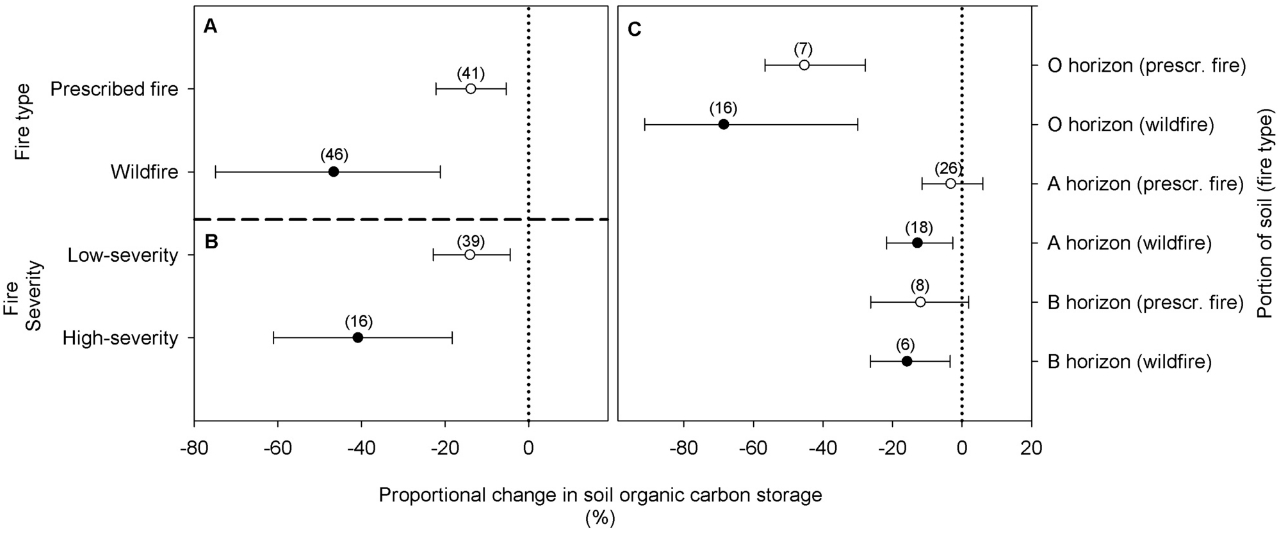

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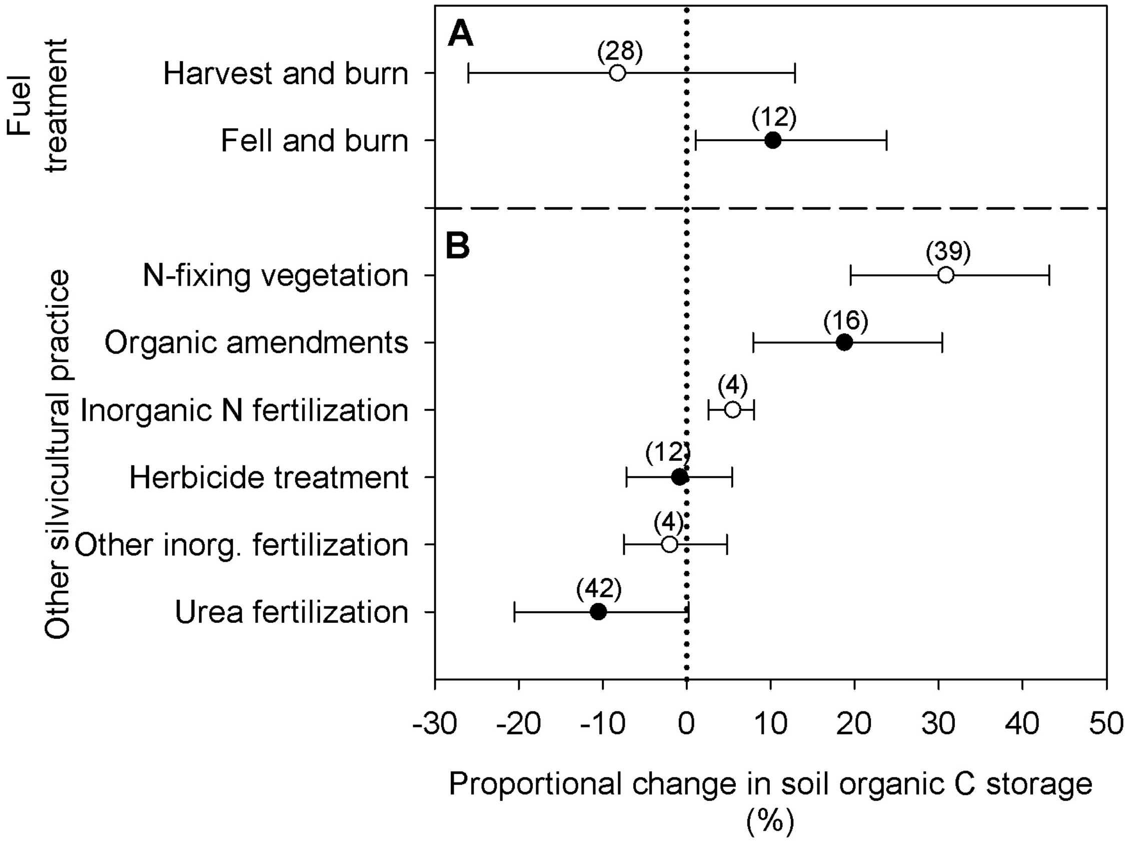

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