5-2019

Losses of mineral soil carbon largely offset biomass accumulation fifteen years after whole-tree harvest in a northern hardwood forest

Matthew A. Vadeboncoeur
University of New Hampshire, Durham, matt.vad@unh.edu

Steven P. Hamburg
Environmental Defense Fund

Chris E. Johnson
Syracuse University

Jonathan Sanderman
Woods Hole Research Center

Recommended Citation

Hamburg SP, Vadeboncoeur MA, Johnson CE, Sanderman J. 2019. Losses of mineral soil carbon largely offset biomass accumulation fifteen years after whole-tree harvest in a northern hardwood forest. Biogeochemistry 144: 1-14. https://dx.doi.org/10.1007/s10533-019-00568-3

Losses of mineral soil carbon largely offset biomass accumulation fifteen years after whole-tree harvest in a northern hardwood forest

Steven P. Hamburg
Center for Environmental Studies
Brown University
Box 1943
Providence, RI 02912
Current address: Environmental Defense Fund
New York, NY 10010
shamburg@edf.edu

Matthew A. Vadeboncoeur *(corresponding author)*
Earth Systems Research Center
University of New Hampshire
8 College Road
Durham, NH 03824
matt.vad@unh.edu
phone: 603-862-4448

Chris E. Johnson
Dept. of Civil and Environmental Engineering
151 Link Hall
Syracuse University
Syracuse, NY 13244

Jonathan Sanderman
Woods Hole Research Center
149 Woods Hole Road
Falmouth, MA 02540

Note: This is a post-peer-review, pre-copyedit version of an article published in *Biogeochemistry*, made available in accordance with Springer's author rights policy. The final authenticated version is available online at:
http://dx.doi.org/10.1007/s10533-019-00568-3

This document should be cited as:
Hamburg SP, Vadeboncoeur MA, Johnson CE, Sanderman J. 2019. Losses of mineral soil carbon largely offset biomass accumulation fifteen years after whole-tree harvest in a northern hardwood forest. *Biogeochemistry* 144: 1-14. doi:10.1007/s10533-019-00568-3

Key words: soil organic matter, bioenergy, forest management, quantitative soil pits, Hubbard Brook Experimental Forest

Acknowledgements

This work was funded by National Science Foundation grants BSR8316953, BSR8717570, 9211768, 9810221, 0423259, and 1114804. Art Johnson, Tom Huntington, Doug Ryan, and Tom Siccama were integral to the design and execution of this long-term study; Jennifer Atlee assisted with the analysis. We also thank the many field and lab assistants who excavated and processed samples from hundreds of soil pits. This work is a contribution to the Hubbard Brook Ecosystem Study (www.hubbardbrook.org). The Hubbard Brook Experimental Forest is an NSF-funded Long-term Ecological Research site (www.lternet.edu), and is operated by the USDA Forest Service Northern Research Station, Newtown Square, PA.

Abstract

Changes in soil carbon stocks following forest harvest can be an important component of ecosystem and landscape-scale C budgets in systems managed for bioenergy or carbon-trading markets. However, these changes are characterized less often and with less certainty than easier-to-measure aboveground stocks. We sampled soils prior to the whole-tree harvest of Watershed 5 at the Hubbard Brook Experimental Forest in 1983, and again in years 3, 8, and 15 following harvest. The repeated measures of total soil C in this stand show no net change in the O horizon over 15 years, though mixing with the mineral soil reduced observed O horizon C in disturbed areas in post-harvest years 3 and 8. Mineral soil C decreased by 15% (20 Mg ha^{-1}) relative to pre-harvest levels by year 8, with no recovery in soil C stocks by year 15. Proportional changes in N stocks were similar. The loss of mineral soil C offset two-thirds of the C accumulation in aboveground biomass over the same 15 years, leading to near-zero net C accumulation post-harvest, after also accounting for the decomposition of slash and roots. If this result is broadly representative, and the extent of forest harvesting is expanded to meet demand for bioenergy or to manage ecosystem carbon sequestration, then it will take substantially longer than previously assumed to offset harvest- or bioenergy-related carbon dioxide emissions with carbon uptake during forest regrowth.

Introduction

Growing interest in using land management as a greenhouse gas mitigation tool through increased carbon storage and the production of bioenergy (Fahey et al. 2010; Raciti et al. 2012; Griscom et al. 2017; National Academies of Sciences Engineering and Medicine 2018) has further heightened the need for an improved understanding of the effects of forest management on soil C (Watson et al. 2000; Nave et al. 2010; Buchholz et al. 2013). This effort requires both reliable baseline data that can be applied on a regional basis, as well as detailed long-term studies of changes in soil carbon stocks under a variety of management scenarios. However, the effects of anthropogenic disturbance on soil carbon stocks in forest ecosystems are more difficult to assess with accuracy and precision than are aboveground stocks (Lawrence et al. 2012). The slow rate at which soil carbon stocks change, and their inherently large spatial variability, make sufficiently large and long-term data sets expensive and difficult to obtain. However, these limitations are not insurmountable, and addressing the question is critical to our understanding of the role that forest soils might play in creating negative emissions (National Academies of Sciences Engineering and Medicine 2018; Nave et al. 2018).

In the northeastern United States, interest in using forest biomass as a source of energy hinges on the idea that such use is “low carbon” or even “carbon neutral”, resulting in little net increase in atmospheric carbon over the medium to long term (Malmsheimer et al. 2008; Richter et al. 2009). Because burning green (fresh) wood yields less energy per mole of carbon than conventional fossil fuels, such energy systems incur a “carbon debt” when regional harvest rates increase (Fargione et al. 2008; Searchinger et al. 2009; Walker et al. 2010; Fanous and Moomaw 2018). If utilizing more bioenergy involves increased harvesting it will require a better understanding of the implications for total ecosystem carbon and in turn on the net release of terrestrial carbon into the atmosphere over time. While the effects of increased harvesting on aboveground biomass accumulation can be effectively estimated using well established accumulation curves (e.g. Heath et al., 2010; Fahey et al., 2010), long-term changes in soil carbon following whole-tree or stem-only harvesting are less well-characterized, especially in mineral soil horizons below 20 cm depth, and must be better understood to calculate with confidence the carbon consequences of significantly increasing forest biomass use for energy production. Deeper soils are often not monitored due to the effort involved in sampling, and the assumption that mineral soil C is old and stable. However, recent work (Diochon and Kellman 2008, 2009; Diochon et al. 2009; Harrison et al. 2011; Lorenz et al. 2011; James and Harrison 2016; Bowd et al. 2019) suggests that deeper soils, which often represent the bulk of forest ecosystem C stocks, may respond more dramatically to ecosystem disturbance than do shallower soils. Sampling depth is one of many methodological differences (Lawrence et al. 2012; Vadeboncoeur et al. 2012) that impose a range of potential measurement biases upon already high variation in C content and fluxes within and among ecosystems.

Meta-analyses of soil C changes after harvesting have found a range of responses. Johnson and Curtis (2001) found a mean 18% increase in shallow (A-horizon) soil C after stem-only clear cutting and an average 6% decrease with whole-tree harvesting. Across all these syntheses, it is clear that few studies have yet implemented repeated measurements at the same site for more than a decade. A synthesis of data specifically from Spodosols indicated no significant overall effect of harvest on soil C, with perhaps modest increases in shallow soil C balancing modest losses in deeper soils (Nave et al. 2010). Harvest intensity appeared to have little effect in this meta-analysis though results were highly variable among studies (Nave et al. 2010; Thiffault et al. 2011; James and Harrison 2016). Without detailed knowledge of the implications of spatially- and temporally-variable patterns in forest harvesting, it will not be possible to estimate the likely effects of major shifts in forest management on soil carbon pools on a regional or global scale.

In addition to its importance in the carbon cycle, soil organic matter is also the dominant stock of nitrogen in temperate forest ecosystems (Lovett et al. 2018). Nitrogen dynamics in forest ecosystems following harvest are also a key topic of continuing interest, given the role N plays in limiting primary production in temperate forests (LeBauer and Treseder 2008; Vadeboncoeur 2010), and the dramatic shifts in the relative importance of litter recycling and biotic demand following a harvest (Rastetter et al. 2013; Vadeboncoeur et al. 2014; Lovett et al. 2018; Valipour et al. 2018).

Study Questions

Soil C and N for the first eight years following the whole-tree harvest of watershed 5 (W5) at the Hubbard Brook Experimental Forest (HBEF) in New Hampshire have been previously reported (Johnson 1995; Johnson et al. 1995). Here we update the results to 15 years post-harvest, using a protocol that allowed us to explicitly examine both pre-existing and harvest-induced spatial variability in soil. We test whether total solum C and N content changed over the 15-year post-harvest record, and examine apparent changes in C and N distribution throughout the soil profile.

Several variables may affect the spatial patterning of the change in soil C following forest harvest: 1) the amount and location of logging debris left on site; 2) the intensity and amount of mechanical soil disturbance caused by the harvesting; 3) subsequent treatments; and 4) the pre-existing heterogeneity of soil C concentration, depth, bulk density, and rock volume. Because 93% of aboveground biomass was removed from the harvested area, the effects of variation in the amount of slash left on site should be minimal. There has been no post-harvest disturbance of the watershed, reducing another potentially complicating variable.

Here we explicitly address the role of shallow soil disturbance up to 8 years after harvest, when it was still visible, and collect a large enough sample (60 pits per sample date) to allow for reasonable change detection, even given the high pre-existing heterogeneity of the soils. This dataset allows us to assess the whole-ecosystem C balance following harvest, with potentially important implications for bioenergy carbon debt and landscape-scale C dynamics.

Methods

Site description

The HBEF encompasses 3,160 ha of the White Mountain National Forest in Woodstock, New Hampshire (43°56'N, 71°45'W). The climate is cool, temperate, humid continental. The average temperature in January is -8°C and in July is 19°C. Winter temperatures have increased significantly by about 2ºC since 1955, while summer temperatures have not changed significantly (Campbell et al. 2010; Hamburg et al. 2013). Precipitation averages about 1400 mm annually, of which about 25% falls as snow (Bailey et al. 2003; Campbell et al. 2010). The bedrock of the south-facing experimental watersheds is Rangeley Formation schist (Burton et al. 2000) overlain by up to 3 m of unsorted basal till, which is generally thinner towards the ridgeline. Soils are primarily well-drained Spodosols (coarse-loamy, mixed, frigid, Typic Haplorthods; Huntington et al., 1988). The soil surface has a pit-and-mound microtopography resulting from boulders and fallen trees (Bormann et al. 1970).

Watershed 5 covers 22 ha between 490 and 760 m elevation on a southeast-facing, 22% slope, and was whole-tree harvested in 1983-5, primarily in the winter. Prior to harvest, the forest had a total aboveground biomass of 207 Mg ha^{-1}, dominated by sugar maple (*Acer saccharum* Marsh.), American beech (*Fagus grandifolia* Ehrh.), and yellow birch (*Betula allegheniensis* Michx. f.), transitioning to red spruce (*Picea rubens* Sarg.), balsam fir (*Abies balsamea* L. (Mill.)), and white birch (*Betula papyrifera* Marsh) at the highest elevations (Johnson et al. 1995; Arthur et al. 2001). The northern hardwood forest present at the time of harvesting had established following the selective cutting of spruce between 1880 and 1917 (Bormann et al. 1970), and the 1938 hurricane, which resulted in 20-60% blowdown at HBEF, though damage was spatially patchy (Peart et al. 1992).

Harvest treatment

In winter 1983-4, trees > 5 cm in diameter were cut with a feller-buncher and chainsaws. Trees from the lower two-thirds of the watershed were skidded off-site in the winter of 1983-84. Skidding from the upper third was largely completed in the summer of 1984. Trees from the steepest 12% of the watershed were not removed until the summer of 1985 (Ryan et al. 1992; Johnson et al. 1995). One very steep area (0.8 ha) and a narrow buffer along the watershed boundary were not harvested. In the remaining 19.9

ha, where we sampled, an average of 16 Mg ha^{-1} of slash (8 Mg C ha^{-1}) remained after skidding (Siccama et al. 1994; Johnson et al. 1995). About 65% of the watershed showed signs of disturbance at the soil surface, with no visible O horizon on about 25% of the watershed surface (Ryan et al. 1992).

In 1999, aboveground live biomass on W5 was 58 Mg ha^{-1}, dominated by pin cherry (*Prunus pensylvanica* L. f.), yellow birch, white birch, and beech (Fahey et al. 2005; Cleavitt et al. 2017). In January 1998 a major ice storm affected the region, but there was little damage to the young trees in W5, unlike in the adjacent mature forest (Rhoads et al. 2002).

Soil sampling

The soils were sampled four times: 1983, prior to harvest; 1986, three growing seasons post-harvest; 1991, eight growing seasons post-harvest; and 1998, 15 growing seasons post-harvest. Quantitative soils pits were excavated at 60 sites in each sampling year using a stratified random scheme with six elevation bands, in order to ensure a representative and well-dispersed sample set (Huntington et al. 1988). While time consuming, quantitative pits yield direct estimates of soil mass per unit land area and, hence, carbon stocks in stony soils (Vadeboncoeur et al. 2012). Prior to harvesting, in 1983, four sampling points were identified in each of 60 25-m x 25-m plots. One of these points was excavated in 1983 and the other three, which were in similar micro-topographic positions, were sampled in the later years. The four points in each plot were all within 20 m of one another on the same contour, allowing for comparisons across sampling times. Not all sampling points were relocated following the harvest. When markers could not be found, new locations were selected based the criteria previously used for each site. In 1986 and 1991 the surface of each pit site was characterized as to whether it had been disturbed in the harvesting, but by 1998 this could not be determined accurately. Each of the 1983 pits was assigned to a USDA soil series, as summarized by Huntington et al. (1988).

Each quantitative pit was $0.5m^2$ in area and was excavated to bedrock or the top of the C horizon in five layers: Oi+Oe (Oie), Oa, 0-10 cm, 10-20 cm, 20+ cm. Mineral soil thickness averaged 53 cm (to the C horizon or bedrock), and generally declined with increasing elevation (Johnson et al. 2000). The separation of O horizons from each other and from the mineral soil was based on visual criteria. Excavated O horizon material was returned to the lab where it was air-dried and sieved to 6 mm (1983 and 1986) or 5 mm (1991 and 1998). Mineral soil was sieved to 12.5 mm in the field, then weighed, homogenized, sub-sampled, air-dried, and sieved to 2 mm in the lab. For carbon analysis, the sieved O horizon samples were ground in an intermediate Wiley mill (60 mesh), and oven-dried at 60°C. Sieved mineral soil sub-samples were oven-dried at 105°C and pulverized using a wig-L-bug prior to carbon analysis. All results are expressed on a 105°C oven-dry basis.

Sample analysis

To allow for comparison to earlier studies that reported organic matter rather than C content, loss-on-ignition was determined by ashing 10 g of oven-dried soil at 500°C overnight. These data also allowed an independent check of the measured C concentrations (C should be approximately half of loss-on-ignition in these non-calcareous soils; Huntington et al. 1989). We used this check to identify samples that may have been insufficiently homogenized before C and N analysis so they could be re-analyzed.

A 2 to 50 mg sub-sample of ground, oven-dried soil was placed in a tin capsule and combusted in modified-Dumas elemental analyzers. The amount of sub-sample varied by layer and with instrument used (CE Instruments NA-1500 in 1983, 1986, 1998 and EA-1108 in 1991). In order to ensure consistency among years, forty-two samples were randomly chosen from the three earlier sampling dates, and were rerun simultaneously in 1999 on a CE Instruments NC2100 analyzer. The reanalysis of a subset of the soils from five pits collected at each of the four sampling efforts (1983, 1986, 1991 and 1998) showed no systematic differences. The re-measured C concentrations in 1998 were on average 3% greater than the original values (R^2=0.93). Reference materials (peach and pine leaves) resulted in a coefficient of variation $\leq$ 1% for both types of samples (n=32 & 38). In addition, 30 archived samples were reanalyzed in 2010 on a Costech 4010 Elemental Analyzer, where data were missing, or to correct or validate data with unusually high or low %C, %N, or C/LOI values. This instrument had been previously validated against the NC2100 instrument. Mass, LOI, and C and N concentration data for each sample are archived online at the LTER data portal.

Analysis of soil data

Estimates of soil C per unit area in each layer in each pit were calculated by multiplying the C concentration for the layer by the oven-dry < 2-mm mass of that layer. Lost data and mislabeled samples (6 out of about 1200 soil samples) were excluded from all subsequent analysis. In 1983, the first pit excavated was sampled by horizon rather than by depth, and in 1986 one pit was excavated in the wrong location. Both of these pits were excluded from the analysis. For two 1998 Oie samples that were lost before C and N analysis, but not before measuring LOI, we estimated %C and %N based on a regression of all 1998 Oie data against LOI (R^2 = 0.74 for C and 0.49 for N).

We excluded from analysis soil samples we identified as outliers using criteria designed to identify only those data that were greatly outside of the range of the rest of the data set; some of these may represent errors in field processing or recording of data. Given the large number of individuals necessary to do the field work, errors were inevitable, despite a concerted effort to double-check all entries. A soil sample was considered an outlier if its C or N content fell outside of three standard deviations of either the yearly average (n = 60) or the combined 4 year average (n = 240) for that layer. This analysis resulted in the exclusion of data from 24 layers (2% of all data), 16 of which were from the O horizon. Exclusions and corrections described above led to small differences between the data reported here and those reported previously for 1983, 1986, and 1991 by Huntington et al., (1988, 1989); Johnson et al., (1991, 1995), and Johnson (1995).

Other C budget data

To place the observed changes in soil C into an ecosystem budget context, we assembled data on other ecosystem C stocks. Where necessary, data from Watershed 6 (pre-harvest), and Watershed 2 (post-harvest) are used. Aboveground biomass was directly measured during the harvest, and reported by Arthur et al. (2001); biomass calculated from post-harvest tree inventories was reported by Fahey et al. (2005). Pre-harvest woody debris is from measurements on watershed 6 in 1978 (Fahey et al. 2005); post-harvest woody debris (i.e. slash) was reported by Siccama (2000), and decayed at approximately 10% annually (see also Arthur et al., 1993; Shortle and Smith, 2015). Stumps were estimated as 2.5% of pre-harvest aboveground biomass (Chojnacky et al. 2014), and conservatively assumed to decay at 5% annually, given the persistence of spruce stumps observed by Bormann et al. (1970).

Root biomass was estimated as the sum of several pools. Pre-harvest live fine and coarse roots were reported by Fahey et al. (1988). Fine roots were assumed to be killed in the harvest and decay exponentially at 25% annually (Fahey et al. 1988), but live fine root biomass was assumed to recover within 10 years of harvest (Park et al. 2007). Coarse roots surviving the harvest (due to root- and stump sprouting; Fahey et al. 1988) were estimated from the coarse root biomass in excess of allometric estimates, observed in ~15-year-old stands at Bartlett Experimental Forest by Vadeboncoeur et al. (2007). The balance of coarse roots was assumed to have died and decayed at an exponential rate of 10% annually (Fahey et al. 1988; Fahey and Arthur 1994). All biomass values were multiplied by 0.50 to estimate C content.

Statistics

We used two statistical approaches to compare C stocks over time: 1) a paired approach that treated pits from the same plot as repeated measures, and 2) an unpaired analysis that treated each pit as an independent observation. We used an unpaired approach to determine the effects of harvest-associated soil disturbance, based on visual observations of disturbance from each year (1986 and 1991). Since in 1998 it was difficult to determine which pits had been disturbed during the harvesting, given 14 years of litterfall, no distinction was made between disturbed and undisturbed pits in 1998.

Student's two-sample *t*-test was used to test for significant differences in pre- and post-harvest soil attributes: loss-on-ignition, organic matter, soil oven-dried mass, C concentration. Paired *t*-tests were used to compare soil C pool sizes between pre- and post-harvest pools within the initially paired pits. As noted by Johnson et al. (1995) the experimental design employed in this study includes no replication on the watershed level and thus suffers from spatial psuedoreplication. The use of spatially paired pits addresses this issue in part, but the lack of multiple randomly-assigned harvested stands, a byproduct of the enormous cost of conducting the experiment, remains unaddressed. Formally, the statistical tests performed only show differences between means for the study watershed across the sampled years, yet we believe that whole-tree harvesting and forest recovery are the most important factors driving these differences.

Results

Changes in C and N stocks at the watershed scale

The total C content of the soil declined significantly over 15 years of post-harvest observations (Table 1; Fig. 1). The C content of the O horizon declined by ~30% over the first eight years post-harvesting, but returned to pre-harvest levels after 15 years (Table 1; Fig. 1). The C content of the mineral soil, by contrast, increased non-significantly by year three, then significantly decreased by year eight, and remained at this lower level in year 15 (Table 1; Fig. 1). The net decrease in observed mineral soil C was 20 Mg ha^{-1}, equal to 15% of the initial mineral soil C and 13% of the initial total solum C. Most of the decline in soil C content occurred below 10 cm depth, though net declines were not significant for the 10-20 cm and 20+ cm layers individually (Table 1). When the soil C content is examined using the paired approach, the 15-year results are similar, but non-significant (Table 1; Fig. 2). Mineral soil N content declined significantly by a proportion similar to that of C (-14% over 15 years). Over the same time span, O horizon N increased non-significantly (by 7%), while C decreased non-significantly (by 5%). The O horizon C/N ratio appears to have declined significantly, from 22.0 to 20.1, over the 15-year study, but no such change is apparent in the mineral soil.

Mineral soil mass did not differ significantly among sampling times (Table 1), though there was a 10% difference between the 1998 and 1986 values, the largest difference observed. There was a significant decrease in the Oa horizon mass between 1986 and 1991, but there were no significant differences between the 1998 masses and those from the other years. There was also a significant increase in the mass of the 0-10 cm layer in the first three years after the harvest, which is related to an increase in bulk density, probably due to compaction by logging machinery (Johnson et al., 1991).

Effects of surface disturbance on C stocks in 1986 and 1991

The effect of surface disturbance on the distribution of soil C after harvesting on W5 was dramatic. O horizon C content on disturbed sites declined by 75% three years following harvesting, while undisturbed pits showed no significant change (Fig. 2b). The C content in mineral soils may have increased more in the disturbed pits (16%) than in the undisturbed pits (5%; Fig. 2b-c), but both increases were non-significant. Across all pits in 1986, the (non-significant) decrease in O horizon C was more than offset by an increase in mineral soil C (Table 1). Eight years after the harvest (1991), O horizon C content remained significantly lower in those pit locations noted as disturbed at the time the pits were dug (Figure 2b).

Changes in C and N across elevation / vegetation zones

The initial soil C distribution varied by vegetation zone; the spruce-fir zone had more C in the O horizon and less in the mineral soil than either of the two hardwood zones. However, there were few significant changes over time in either disturbance class in any zone though there were differences between years as large as 81% (unpaired *t*-tests; Table 2). Only the lower hardwood zone had a significant net decrease in mineral soil C in 1998 (20% = 27 Mg C ha^{-1}).

Discussion

Harvest-associated disturbance effects on O horizon C and N

In post-harvest pits that were noted as visibly disturbed, which represented about one-third of sampled locations, O horizon material was pushed aside and/or mixed into mineral soil by logging machinery. However, it is not clear to what degree this disturbance resulted in a net loss of soil C from the system, as the C content of the mineral soil in the disturbed pits in 1986 may (given the uncertainty) have increased by enough to fully explain the significant O horizon loss by mixing, although this was not the case in 1991. The data illustrate that shallow, horizon-based sampling alone is insufficient in situations where horizons are altered by disturbance, and that the disturbance effects of a harvest potentially confuse comparisons between pre-harvest and post-harvest samples (Johnson et al. 1991; Ryan et al. 1992). Overall, C storage in the O horizon recovered completely by 1998. In the adjacent reference watershed (W6), which was not cut, there was no significant change in O horizon mass or C content over the same time period (Fahey et al. 2005; Yanai et al. 2013).

Using a chronosequence approach, Covington (1981) and Federer (1984) deduced an apparent decline in O horizon mass of approximately 50% in the first two decades following harvesting of northern hardwoods. Repeated sampling at Federer's stands did not confirm this observation (Yanai et al. 2000, 2003), a discrepancy attributed to changes in disturbance intensity with changes in harvest intensity, equipment, and technique over the 20th Century. Quantitative pit data from a chronosequence of sites, which shares some sites with Federer's chronosequence, also showed no clear pattern in either mineral soil or O horizon C content with forest age, though sample size was small (3 pits per site; Vadeboncoeur et al., 2012).

O horizon C:N ratio appears to have decreased somewhat (Table 1), but no such change was observed in the mineral soils. In W6 (an adjacent, unmanipulated reference), the O horizon C:N ratio has significantly increased during the same time period (Yanai et al., 2013), so the significant decrease in C:N at W5 is likely attributable to a treatment effect. Early-successional species such as pin cherry and birches have high %N relative to the later-successional species mix they replaced (Likens and Bormann 1970), which may explain the shift. Mixing of soil horizons cannot explain the 15-yr net change, since the disturbed pits had higher O-horizon C:N when analyzed separately. Between 1983 and 1986, O horizon C:N increased, especially in disturbed pits, likely due to the input of dead roots and woody litter.

Organic matter loss at depth

Unlike the O horizon, which did not appear to lose C except where physically disturbed by the harvest, mineral soil C content declined significantly between 1983 and 1991, and remained significantly lower than pre-harvest through 1998 (Table 1). This finding confirms a non-significant observation reported by Johnson (1995) based on a slightly different handling of the 1991 data. This loss is not significant for any single layer, but quantitatively appears to come mostly from the 10-20 and 20+ cm increments. Shallow disturbance associated with the harvest may explain the lack of a significant response in the 0-10 cm horizon. Any losses of C from this layer may be balanced by mixing of O horizon material into the mineral soil at disturbed sites. Sampling error in soil mass does not explain the observed trend in C content; there are no significant differences in mineral soil mass, depth, and coarse fragment volume among years.

Our results differ somewhat from the meta-analysis results of Nave et al. (2010) for post-harvest Spodosols, in which modest increases in C pools in surface horizons approximately balanced deeper losses, though James and Harrison (2016) report that Spodosol organic and mineral soils both lost C post-harvest. Harvesting intensity may explain some of the differences among studies – the high intensity of the W5 harvest left relatively little slash to contribute to the post-harvest O horizon. Diochon et al. (2009) showed large apparent deep-soil losses of C in a chronosequence of spruce clearcuts in Nova Scotia; perhaps as much as 50% of total solum C over the first 30 years of regeneration, mostly in deeper horizons. Examination of carbon isotope ratios of different physical C fractions (Diochon and Kellman, 2009) added evidence for increased mineralization of organic matter below 20 cm in young forests (< 45 years). Zummo and Friedland (2011) observed similar apparent isotopic differences between harvested and reference stands at Bartlett Experimental Forest only three years post-harvest. However, as noted earlier, chronosequences that lack replication within age or disturbance classes have the potential to lead to spurious conclusions due to unquantified pre-existing site differences, so they should be interpreted with caution. Along similar lines, the distribution of SOC was observed to have shifted towards less-recalcitrant forms in Oa, E, and Bh horizons in horizon-based samples from a subset of the W5 soil pits (Ussiri and Johnson 2007), while most of the apparent C loss we observed was below 10 cm, in the Bs1 and Bs2 horizons.

Changes in total soil C and N stocks

Across the harvested watershed, the net loss of soil C 15 years post-harvest was about 20 Mg C ha^{-1}, equal to 15% of the pre-harvest mineral soil C stock (Fig. 1; Table 1). For comparison, the direct C impact of the harvest was the removal of 90 Mg C ha^{-1} as aboveground biomass, of which about 30 Mg C ha^{-1} had been replaced by 1999 (Fahey et al. 2005). Including changes in total solum C in an ecosystem budget (Fig. 3) shows the ecosystem as a possible net source of C (losing 11 Mg C ha^{-1} from 1984-98; though note that the SE on the change in total soil C was 10 Mg C ha^{-1}) in the first 15 years post-harvest, despite rapidly aggrading aboveground biomass.

Despite the large number of pits excavated in this study, the overall ecosystem C budget retains a high level of uncertainty. The longevity of C in slash and dead coarse roots after harvesting, and the degree to which their residues contribute to the pools now sampled as "soil" C is somewhat unclear, though the decay of such coarse woody material is certainly slow (Arthur et al. 1993; Fahey and Arthur 1994). Total soil respiration at W5 was elevated relative to the reference watershed in years 6-8 post-harvest, but not in years 12-13 (Fahey et al. 2005). If harvest residues contribute to the recovery or maintenance of soil C content post-harvest, whole-tree harvests for bioenergy must be regarded as fundamentally different from bole-only harvests. Belowground C inputs are also important to consider, as the slow decay of root systems killed by the harvest may contribute substantially to soil organic matter over time. These effects may vary by species and silvicultural treatment, as not all root systems are necessarily killed by the harvest; for example, young stands with strong vegetative regeneration of beech and red maple retain much of the pre-harvest live root biomass (Vadeboncoeur et al. 2007).

Management and policy implications

The observed apparent decline in mineral soil C in the first decade (or longer; Bowd et al., 2019; Diochon et al., 2009) following forest harvest affects net landscape greenhouse gas emissions from logging if recovery of that soil carbon takes longer than the interval between logging events, or if the amount of land being harvested increases over time. Longer-term studies will be required to determine the rate at which soil C re-equilibrates to pre-harvest values, or indeed whether it does at management-relevant time scales. Also relevant are the soil C impacts of more frequent partial cutting regimes which supply much of the bioenergy wood demand in the northeastern U.S. (Littlefield and Keeton 2012).

Buchholz et al. (2013) noted little agreement in the literature on the mechanism explaining observed declines in soil C following harvest. Future studies that can test these mechanisms individually under realistic forest-management scenarios are crucial to understanding not only how but why soil C responds to forest harvest, and may ultimately allow models to predict which types of forests and soils are most susceptible to these losses under various types of management, providing a sound scientific basis for improved management of forest soil C stocks.

The observed losses of mineral soil C substantially reduce the initial rate of net ecosystem C sequestration to near-zero over the first 15 years, but assuming soil C re-equilibrates within each cutting rotation, it does not necessarily indicate a decline in landscape-scale C stocks; current harvest rates appear to be sustainable and at sustained levels across most of the northeastern U.S. (Buchholz et al. 2011). If the distribution of stands of various ages is kept relatively constant as a result of sustained harvesting regimes, total carbon stocks across each regional woodshed should be static, absent other potential soil C drivers (e.g., climate change). The fact that changes in soil C below the traditionally sampled shallow horizons may be quantitatively more important than previously thought presents a challenge for getting the accounting right if harvesting rates increase substantially as a result of the demand for wood products or bioenergy. The lack of change between 1991 and 1998 on W5 (Figure 1) implies that perhaps mineral soil C stocks stabilized after 8 years post-harvest, but further observations will be necessary to validate this interpretation.

Validating our results using repeated measures elsewhere and a better understanding of the mechanisms that are driving the observed changes (e.g. profile mixing, priming, changes in soil moisture and temperature, altered nutrient availability and microbial communities) will require characterization of the forms of carbon that have been lost from the soil. Modeling the trajectory of soil C in a managed forest in a changing climate over multiple rotations will also require a great deal of additional long-term data from W5 and other stands like it. Key to correctly interpreting the implications of our long-term results from W5 is making sure that they are viewed in the context of the larger managed landscape and not as a single stand.

Data accessibility

All original data presented in this paper are available from the LTER Network Data Portal at: http://doi.org/10.6073/pasta/4bad076230659350cfbd9ccf92fc3dcf

References

Arthur MA, Hamburg SP, Siccama TG (2001) Validating allometric estimates of aboveground living biomass and nutrient contents of a northern hardwood forest. Can J For Res 31:11–17. doi: 10.1139/x00-131

Arthur MA, Tritton LM, Fahey TJ (1993) Dead bole mass and nutrients remaining 23 years after clear-felling of a northern hardwood forest. Can J For Res 23:1298–1305. doi: 10.1139/x93-166

Bailey AS, Hornbeck JW, Campbell JL, Eagar C (2003) Hydrometeorological Database for Hubbard Brook Experimental Forest: 1955 – 2000. USDA Forest Service Northern Research Station, General Technical Report GTR NE-305

Bormann FH, Siccama TG, Likens GE, Whittaker RH (1970) The Hubbard Brook Ecosystem Study: Composition and dynamics of the tree stratum. Ecol Monogr 40:373–388. doi: 10.2307/1942336

Bowd EJ, Banks SC, Strong CL, Lindenmayer DB (2019) Long term impacts of wildfire and logging on forest soils. Nat Geosci, in press. doi: 10.1038/s41561-018-0294-2

Buchholz T, Canham CD, Hamburg SP (2011) Forest biomass and bioenergy: opportunities and constraints in the northeastern United States. Cary Institute of Ecosystem Studies, Millbrook, NY. https://www.caryinstitute.org/sites/default/files/public/downloads/news/report_biomass.pdf

Buchholz T, Friedland AJ, Hornig CE, et al (2013) Mineral soil carbon fluxes in forests and implications for carbon balance assessments. GCB Bioenergy 6:305–311. doi: 10.1111/gcbb.12044

Burton WC, Walsh GJ, Armstrong TR (2000) Bedrock geologic map of the Hubbard Brook Experimental Forest, Grafton County, New Hampshire. USGS Open File Report 00-45

Campbell JL, Ollinger SV, Flerchinger GN, et al (2010) Past and projected future changes in snowpack and soil frost at the Hubbard Brook Experimental Forest, New Hampshire, USA. Hydrol Process 24:2465–2480. doi: 10.1002/hyp.7666

Chojnacky DC, Heath LS, Jenkins JC (2014) Updated generalized biomass equations for North American tree species. Forestry 87:129–151. doi: 10.1093/forestry/cpt053

Cleavitt NL, Battles JJ, Johnson CE, Fahey TJ (2017) Long-term decline of sugar maple following forest harvest, Hubbard Brook Experimental Forest, New Hampshire. Can J For Res 48:23–31. doi: 10.1139/cjfr-2017-0233

Covington WW (1981) Changes in forest floor organic matter and nutrient content following clear cutting in northern hardwoods. Ecology 62:41–48. doi: 10.2307/1936666

Diochon AC, Kellman L (2009) Physical fractionation of soil organic matter: Destabilization of deep soil carbon following harvesting of a temperate coniferous forest. J Geophys Res 114:1–9 . doi: 10.1029/2008JG000844

Diochon AC, Kellman L (2008) Natural abundance measurements of ^{13}C indicate increased deep soil carbon mineralization after forest disturbance. Geophys Res Lett 35:L14402. doi: 10.1029/2008GL034795

Diochon AC, Kellman L, Beltrami H (2009) Looking deeper: An investigation of soil carbon losses following harvesting from a managed northeastern red spruce (Picea rubens Sarg.) forest chronosequence. For Ecol Manage 257:413–420. doi: 10.1016/j.foreco.2008.09.015

Fahey TJ, Arthur MA (1994) Further studies of root decomposition following harvest of a northern hardwoods forest. For Sci 40:618–629. doi: 10.1093/forestscience/40.4.618

Fahey TJ, Hughes JW, Pu M, Arthur MA (1988) Root decomposition and nutrient flux following whole-tree harvest of northern hardwood forest. For Sci 34:744–768. doi: 10.1093/forestscience/34.3.744

Fahey TJ, Siccama TG, Driscoll CT, et al (2005) The biogeochemistry of carbon at Hubbard Brook. Biogeochemistry 75:109–176. doi: 10.1007/s10533-004-6321-y

Fahey TJ, Woodbury PB, Battles JJ, et al (2010) Forest carbon storage: ecology, management, and policy. Front Ecol Environ 8:245–252. doi: 10.1890/080169

Fanous J, Moomaw WR (2018) A Critical Look at Forest Bioenergy: Exposing a high carbon "climate solution." Somerville, MA. Global Development and Environment Institute, Tufts University, Climate Policy Brief 8. http://ase.tufts.edu/gdae/Pubs/climate/ClimatePolicyBrief8.pdf

Fargione J, Hill J, Tilman D, et al (2008) Land clearing and the biofuel carbon debt. Science 319:1235–1238 . doi: 10.1126/science.1152747

Federer CA (1984) Organic matter and nitrogen content of the forest floor in even-aged northern hardwoods. Can J For Res 14:763–767. doi: 10.1139/x84-136

Griscom BW, Adams J, Ellis PW, et al (2017) Natural climate solutions. Proc Natl Acad Sci 114:11645–11650. doi: 10.1073/pnas.1710465114

Hamburg SP, Vadeboncoeur MA, Richardson AD, Bailey AS (2013) Climate change at the ecosystem scale: a 50-year record in New Hampshire. Clim Change 116:457–477. doi: 10.1007/s10584-012-0517-2

Harrison RB, Footen PW, Strahm BD (2011) Deep soil horizons: contribution and importance to soil carbon pools and in assessing whole-ecosystem response to management and global change. For Sci 57:67–76. doi:10.1093/forestscience/57.1.67

Heath LS, Nichols MC, Smith JE, Mills JR (2010) FORCARB2: an updated version of the U. S. forest carbon budget model. USDA Forest Service Northern Research Station, General Technical Report GTR-67.

Huntington TG, Johnson CE, Johnson AH, et al (1989) Carbon, organic matter, and bulk density relationships in a forested spodosol. Soil Sci 148:380–386

Huntington TG, Ryan DF, Hamburg SP (1988) Estimating Soil Nitrogen and Carbon Pools in a Northern Hardwood Forest Ecosystem. Soil Sci Soc Am J 52:1162–1167. doi: 10.2136/sssaj1988.03615995005200040049x

James J, Harrison R (2016) The Effect of Harvest on Forest Soil Carbon: A Meta-Analysis. Forests 7:308. doi: 10.3390/f7120308

Johnson CE (1995) Soil nitrogen status 8 years after whole-tree clear-cutting. Can J For Res 25:1346–1355. doi: 10.1139/x95-147

Johnson CE, Driscoll CT, Fahey TJ, et al (1995) Carbon dynamics following clear-cutting of a northern hardwood forest. In: McFee WW, Kelly JM (eds) Carbon forms and functions in forest soils. SSSA, Madison, WI, pp 463–488

Johnson CE, Driscoll CT, Siccama TG, Likens GE (2000) Element Fluxes and Landscape Position in a Northern Hardwood Forest Watershed Ecosystem. Ecosystems 3:159–184. doi: 10.1007/s100210000017

Johnson CE, Johnson AH, Huntington TG, Siccama TG (1991) Whole-Tree Clear-Cutting Effects on Soil Horizons and Organic-Matter Pools. Soil Sci Soc Am J 55:497–502. doi: 10.2136/sssaj1991.03615995005500020034x

Johnson DW, Curtis PS (2001) Effects of forest management on soil C and N storage: meta analysis. For Ecol Manage 140:227–238. doi: 10.1016/S0378-1127(00)00282-6

Lawrence GB, Shortle WC, David MB, et al (2012) Early indications of soil recovery from acidic deposition in US red spruce forests. Soil Sci Soc Am J 76:1407–1417. doi: 10.2136/sssaj2011.0415

LeBauer DS, Treseder KK (2008) Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. Ecology 89:371–379. doi: 10.1890/06-2057.1

Likens GE, Bormann FH (1970) Chemical analyses of plant tissues from the Hubbard Brook ecosystem in New Hampshire. Yale University School of Forestry, Bulletin 79.

Littlefield CE, Keeton WS (2012) Bioenergy harvesting impacts on ecologically important stand structure and habitat characteristics. Ecol Appl 22:1892–1909. doi: 10.1890/11-2286.1

Lorenz K, Lal R, Shipitalo MJ (2011) Stabilized soil organic carbon pools in subsoils under forest are potential sinks for atmospheric CO_2. For Sci 57:19–25. doi: 10.1093/forestscience/57.1.19

Lovett GM, Goodale CL, Ollinger S V, et al (2018) Nutrient retention during ecosystem succession: a revised conceptual model. Front Ecol Environ 16:532–538. doi:

10.1002/fee.1949

Malmsheimer RW, Heffernan P, Brink S, et al (2008) Forest management solutions for mitigating climate change in the United States. J For 106:115–171. doi: 10.1093/jof/106.3.115

National Academies of Sciences Engineering and Medicine (2018) Negative Emissions Technologies and Reliable Sequestration. National Academies Press, Washington, D.C. https://www.nap.edu/catalog/25259

Nave LE, Domke GM, Hofmeister KL, et al (2018) Reforestation can sequester two petagrams of carbon in US topsoils in a century. Proc Natl Acad Sci 115:201719685. doi: 10.1073/pnas.1719685115

Nave LE, Vance ED, Swanston CW, Curtis PS (2010) Harvest impacts on soil carbon storage in temperate forests. For Ecol Manage 259:857–866. doi: 10.1016/j.foreco.2009.12.009

Park BB, Yanai RD, Vadeboncoeur MA, Hamburg SP (2007) Estimating root biomass in rocky soils using pits, cores, and allometric equations. Soil Sci Soc Am J 71:206–213. doi: 10.2136/sssaj2005.0329

Peart DR, Cogbill CV, Palmiotto PA (1992) Effects of logging history and hurricane damage on canopy structure in a northern hardwoods forest. Bull Torrey Bot Club 119:29–38. doi: 10.2307/2996917

Raciti SM, Fahey TJ, Thomas RQ, et al (2012) Local-scale carbon budgets and mitigation opportunities for the northeastern United States. Bioscience 62:23–38. doi: 10.1525/bio.2012.62.1.7

Rastetter EB, Yanai RD, Thomas RQ, et al (2013) Recovery from disturbance requires resynchronization of ecosystem nutrient cycles. Ecol Appl 23:621–642. doi: 10.1890/12-0751.1

Rhoads AG, Hamburg SP, Fahey TJ, et al (2002) Effects of an intense ice storm on the structure of a northern hardwood forest. Can J For Res 32:1763–1775. doi: 10.1139/X02-089

Richter DD, Jenkins DH, Karakash JT, et al (2009) Wood energy in America. Science (80-) 323:1432–1433. doi: 10.1126/science.1166214

Ryan DF, Huntington TG, Martin CW (1992) Redistribution of soil nitrogen, carbon and organic matter by mechanical disturbance during whole-tree harvesting in northern hardwoods. For Ecol Manage 49:87–99. doi: 10.1016/0378-1127(92)90162-3

Searchinger TD, Hamburg SP, Melillo J, et al (2009) Fixing a critical climate accounting error. Science (80-) 326:527–528. doi: 10.1126/science.1178797

Shortle W, Smith K (2015) Wood Decay Fungi Restore Essential Calcium to Acidic Soils in Northern New England. Forests 6:2571–2587. doi: 10.3390/f6082571

Siccama TG (2000) Watershed 5 Whole-Tree Harvest: Baseline and Initial Results. In: Hubbard Brook Ecosystem Study. http://hubbardbrook.org/watersheds/watershed-5

Siccama TG, Hamburg SP, Arthur MA, et al (1994) Corrections to allometric equations and plant tissue chemistry for Hubbard Brook Experimental Forest. Ecology 75:246–248 . doi: 10.2307/1939398

Thiffault E, Hannam KD, Paré D, et al (2011) Effects of forest biomass harvesting on soil productivity in boreal and temperate forests — a review. Environ Rev 19:278–309. doi: 10.1139/A11-009

Ussiri DAN, Johnson CE (2007) Organic matter composition and dynamics in a northern hardwood forest ecosystem 15 years after clear-cutting. For Ecol Manage 240:131–142. doi: 10.1016/j.foreco.2006.12.017

Vadeboncoeur MA (2010) Meta-analysis of fertilization experiments indicates multiple limiting nutrients in northeastern deciduous forests. Can J For Res 40:1766–1780. doi: 10.1139/X10-127

Vadeboncoeur MA, Hamburg SP, Blum JD, et al (2012) The quantitative soil pit method for measuring belowground carbon and nitrogen stocks. Soil Sci Soc Am J 76:2241–2255. doi: 10.2136/sssaj2012.0111

Vadeboncoeur MA, Hamburg SP, Yanai RD (2007) Validation and refinement of allometric equations for roots of northern hardwoods. Can J For Res 37:1777–1783. doi: 10.1139/X07-032

Vadeboncoeur MA, Hamburg SP, Yanai RD, Blum JD (2014) Rates of sustainable forest harvest depend on rotation length and weathering of soil minerals. For Ecol Manage 318:194–205. doi: 10.1016/j.foreco.2014.01.012

Valipour, M., Driscoll, C.T., Johnson, C.E., Battles, J.J., Campbell, J.L., Fahey, T.J., 2018. The application of an integrated biogeochemical model to simulate dynamics of vegetation, hydrology and nutrients in soil and streamwater following a whole-tree harvest of a northern hardwood forest. Sci. Total Environ. 645, 244–256. doi:10.1016/j.scitotenv.2018.07.066

Walker T, Cardellichio P, Colnes A, et al (2010) Biomass sustainability and carbon policy study. Manomet Center for Conservation Sciences, Manomet, MA. https://www.manomet.org/sites/default/files/publications_and_tools/Manomet_Biomass_Report_Full_June2010.pdf

Watson RT, Noble IR, Bolin B, et al (2000) Land use, land-use change, and forestry. Cambridge University Press, Cambridge, UK

Yanai RD, Arthur MA, Siccama TG, Federer CA (2000) Challenges of measuring forest floor organic matter dynamics: repeated measures from a chronosequence. For Ecol Manage 138:273–283. doi: 10.1016/S0378-1127(00)00402-3

Yanai RD, Stehman S V, Arthur MA, et al (2003) Detecting change in forest floor carbon. Soil Sci Soc Am J 67:1583–1593. doi: 10.2136/sssaj2003.1583

Yanai RD, Vadeboncoeur MA, Hamburg SP, et al (2013) From missing source to missing sink: Long-term changes in the nitrogen budget of a northern hardwood forest. Environ Sci Technol 47:11440–11448. doi: 10.1021/es4025723

Zummo LM, Friedland AJ (2011) Soil carbon release along a gradient of physical disturbance in a harvested northern hardwood forest. For Ecol Manage 261:1016–1026. doi: 10.1016/j.foreco.2010.12.022

Figure 1. C stocks in the O horizon and mineral soil pre-harvest (1983) through 15-years post-harvest (1998). Pits were not classified as disturbed or undisturbed in 1998. Error bars show 1 SE. *** indicates significant differences between disturbed and undisturbed pits at the p <0.001 level.

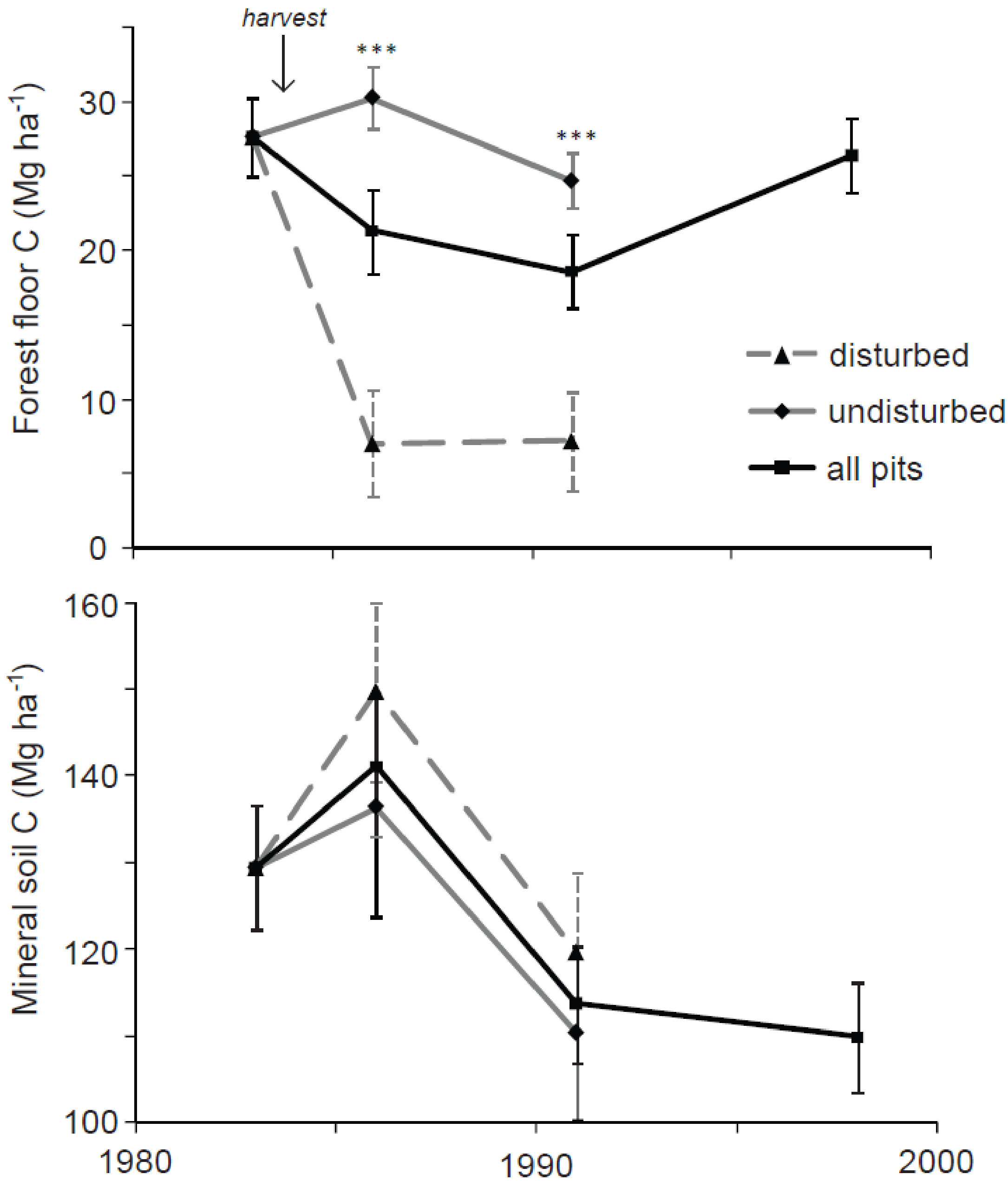

Figure 2. Net changes in C content from pre-harvest. Significant changes are denoted as follows: * indicates $p < 0.05$; *** indicates $p < 0.001$.

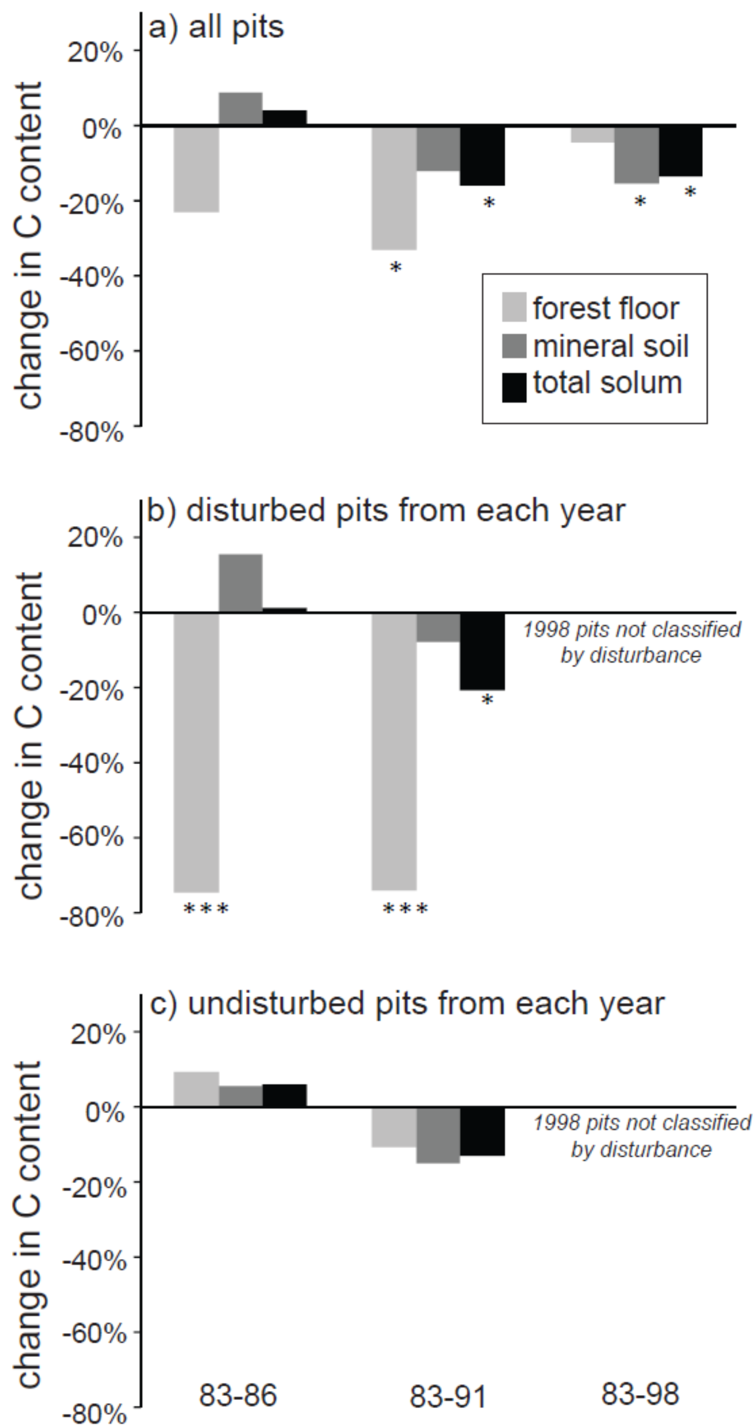

Figure 3: Total ecosystem C stocks at Watershed 5 in the first 15 years post-harvest.

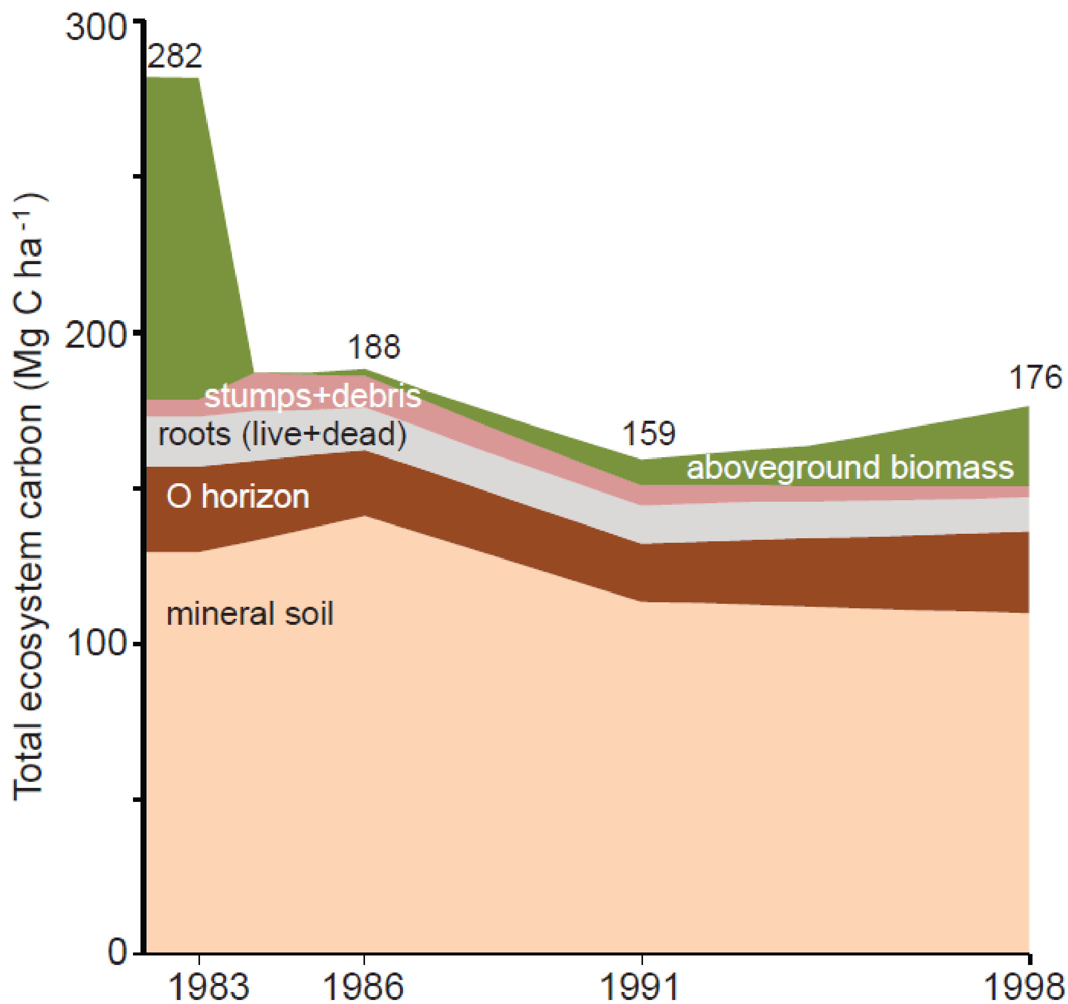

Table 1. Summary statistics for C, N, soil mass, and organic matter over 15 years. Across years, mean values that do not share a superscript are significantly different (unpaired *t*-test $p < 0.05$). Standard errors are shown in italics. Asterisks indicate significant 15-year changes for plots with non-excluded pits in both 1983 and 1998 (paired t-test). MS = mineral soil.

Layer	1983		1986		1991		1998		15-yr change in paired pits	
					soil C (Mg ha^{-1})					
O_{ie}	9.6 [a]	*0.7*	6.6 [b]	*0.8*	6.5 [b]	*0.8*	7.4 [b]	*0.5*	-2.2 *	*0.9*
O_a	18.9 [a]	*2.3*	15.7 [a]	*2.4*	13.4 [a]	*2.2*	19.3 [a]	*2.3*	1.5	*2.8*
0-10 cm	32.2 [a]	*1.1*	35.3 [a]	*1.4*	31.6 [a]	*1.5*	32.4 [a]	*1.4*	-0.1	*1.7*
10-20 cm	26.5 [ab]	*1.7*	29.6 [a]	*1.5*	23.0 [b]	*1.4*	24.5 [b]	*1.4*	-2.2	*2.2*
20+ cm	69.4 [ab]	*6.1*	74.7 [a]	*6.4*	59.9 [ab]	*5.4*	53.8 [b]	*5.3*	-13.1	*8.4*
O horizon	27.7 [ac]	*2.7*	21.3 [abc]	*2.8*	18.6 [b]	*2.5*	26.4 [c]	*2.6*	0.1	*3.2*
MS	129.4 [ab]	*7.2*	141.1 [a]	*8.0*	113.7 [bc]	*6.8*	109.8 [c]	*6.3*	-18.4	*9.9*
Solum	158.4 [a]	*7.6*	164.9 [a]	*7.6*	133.6 [b]	*7.2*	137.4 [b]	*6.8*	-18.5	*9.5*
					soil N (kg ha^{-1})					
O_{ie}	431 [a]	*32*	293 [b]	*35*	321 [b]	*36*	369 [ab]	*24*	-62	*41*
O_a	837 [ab]	*96*	711 [ab]	*100*	625 [a]	*97*	979 [b]	*119*	194	*131*
0-10 cm	1595 [a]	*62*	1630 [a]	*67*	1584 [a]	*70*	1567 [a]	*69*	-42	*84*
10-20 cm	1176 [a]	*72*	1249 [a]	*68*	1100 [a]	*65*	1103 [a]	*66*	-79	*100*
20+ cm	2941 [ab]	*243*	3034 [a]	*260*	2843 [ab]	*250*	2312 [b]	*224*	-540	*335*
O horizon	1238 [ac]	*113*	971 [ab]	*122*	903 [b]	*116*	1329 [c]	*130*	146	*156*
MS	5747 [ab]	*302*	5971 [a]	*337*	5502 [ab]	*319*	4937 [b]	*271*	-809 *	*406*
Solum	7020 [a]	*296*	7110 [a]	*332*	6483 [a]	*334*	6363 [a]	*290*	-590	*391*
					carbon/nitrogen (mass ratio)					
O_{ie}	22.5 [a]	*0.3*	23.2 [a]	*0.5*	19.9 [b]	*0.5*	20.2 [b]	*0.3*	-2.2 ***	*0.5*
O_a	21.4 [a]	*0.4*	21.5 [a]	*0.4*	20.7 [ab]	*0.6*	19.8 [b]	*0.3*	-1.8 **	*0.6*
0-10 cm	20.5 [ac]	*0.3*	21.9 [b]	*0.4*	20.0 [c]	*0.5*	21.0 [abc]	*0.4*	0.5	*0.5*
10-20 cm	22.7 [ad]	*0.5*	24.2 [b]	*0.5*	21.0 [c]	*0.5*	22.4 [d]	*0.4*	-0.4	*0.6*
20+ cm	23.4 [ad]	*0.5*	24.9 [bd]	*0.6*	21.1 [c]	*0.6*	23.2 [d]	*0.4*	0.2	*0.5*
O horizon	22.0 [a]	*0.3*	22.3 [a]	*0.5*	20.4 [b]	*0.5*	20.1 [b]	*0.3*	-1.8 ***	*0.5*
MS	22.5 [ad]	*0.3*	23.9 [b]	*0.4*	20.5 [c]	*0.4*	22.2 [d]	*0.3*	-0.1	*0.4*
Solum	22.4 [a]	*0.3*	23.4 [a]	*0.4*	20.6 [b]	*0.4*	21.5 [c]	*0.2*	-0.9 *	*0.3*
					soil mass (kg m^{-2})					
O_{ie}	2.1 [a]	*0.1*	1.9 [ab]	*0.3*	1.5 [b]	*0.2*	1.7 [ab]	*0.1*	-0.3	*0.2*
O_a	6.4 [ab]	*0.7*	8.4 [a]	*1.4*	5.1 [b]	*0.8*	7.0 [ab]	*0.8*	1.0	*1.0*
0-10 cm	50 [ac]	*2*	58 [b]	*3*	48 [c]	*2*	52 [abc]	*2*	2.6	*2.6*
10-20 cm	52 [a]	*3*	61 [a]	*4*	51 [a]	*3*	55 [a]	*3*	2.7	*4.1*
20+ cm	213 [a]	*18*	218 [a]	*20*	225 [a]	*20*	197 [a]	*22*	-10.8	*25.5*
O horizon	8 [ab]	*0.8*	10 [a]	*1.5*	6 [b]	*0.9*	9 [ab]	*0.9*	0.8	*1.1*
MS	320 [a]	*21*	341 [a]	*22*	324 [a]	*23*	303 [a]	*24*	-11.2	*28.3*
Solum	339 [a]	*22*	365 [a]	*23*	330 [a]	*23*	320 [a]	*26*	1.4	*32.5*
					soil organic matter (Mg ha^{-1})					
O_{ie}	18 [a]	*1.3*	13 [b]	*1.5*	12 [b]	*1.5*	14 [b]	*0.9*	-3.3 *	*1.6*
O_a	32 [a]	*3.8*	30 [a]	*4.6*	23 [a]	*3.9*	34 [a]	*4.2*	4.7	*5.0*
0-10 cm	67 [a]	*6*	65 [a]	*2*	71 [a]	*3*	72 [a]	*6*	4.7	*8.0*
10-20 cm	54 [a]	*3*	57 [a]	*3*	55 [a]	*3*	54 [a]	*3*	-0.3	*4.3*
20+ cm	159 [a]	*14*	157 [a]	*14*	154 [a]	*13*	132 [a]	*13*	-21.5	*19.6*
O horizon	48 [ac]	*5*	42 [abc]	*5*	34 [b]	*5*	48 [c]	*5*	3.6	*5.7*
MS	284 [a]	*17*	282 [a]	*16*	280 [a]	*17*	256 [a]	*16*	-25.1	*24.3*
Solum	338 [a]	*19*	327 [a]	*16*	316 [a]	*16*	305 [a]	*17*	-22.7	*24.8*
					number of pits included in mean (excludes outliers and missing data)					
O_{ie}	51		57		58		58		49	
O_a	56		54		59		57		53	
0-10 cm	58		58		60		60		58	
10-20 cm	59		58		60		59		58	
20+ cm	58		58		59		59		57	
O horizon	53		52		57		55		48	
MS	57		57		59		58		55	
Solum	51		50		57		53		44	

Table 2. Soil C stocks over fifteen years in three vegetation/elevation zones. SF = spruce-fir, HH= high hardwoods, LH = low hardwoods. NR = not reported. Significant incremental and cumulative changes are marked with an asterisk where $p < 0.05$ (unpaired *t*-test).

				soil carbon (Mg ha^{-1})									
		number of pits		O horizon					mineral soil				
elev	year	undisturb	disturb	undisturb	disturb	all	incremental change(%)	cumulative change %	undisturb	disturb	all	incremental change(%)	cumulative change %
SF	1983	13	0			41.7					99		
SF	1986	11	2	42.5	14.0	36.2	-13%	-13%	205	117	132	33%	33%
SF	1991	9	4	32.6	3.6	23.9	-34%	-43%	113	81	92	-30%	-8%
SF	1998	NR	NR	NR	NR	38.0	59%	-9%	NR	NR	102	11%	3%
HH	1983	20	0			20.2					142		
HH	1986	12	8	22.4	6.3	16.0	-21%	-21%	155	158	157	10%	10%
HH	1991	11	9	16.2	6.1	11.6	-27%	-42%	123	131	127	-19%	-11%
HH	1998	NR	NR	NR	NR	21.1	81%*	4%	NR	NR	119	-6%	-16%
LH	1983	27	0			27.4					134		
LH	1986	16	11	30.9	6.2	20.2	-26%	-26%	135	134	135	1%	1%
LH	1991	19	8	26.8	9.9	21.8	8%	-20%	115	111	114	-16%	-15%
LH	1998	NR	NR	NR	NR	26.0	19%	-5%	NR	NR	107	-6%	-20%*