Soil organic carbon storage and soil recovery from acidification in a northern temperate deciduous forest





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Abstract

An understanding of the dynamics of the global carbon cycle is essential in developing plans to mitigate climate change. Approximately two-thirds of total terrestrial carbon is stored in the world's soils and these soil carbon pools exist in a dynamic equilibrium with the surrounding ecosystem. Forest soils in particular have a high density of soil carbon. The capacity of soils to store carbon is impacted by both biotic and abiotic factors. Land-use change and agricultural expansion in particular can greatly diminish the soil carbon stock. Less well understood is the relationship between acid deposition due to acid rain and the dynamics of the soil organic carbon (SOC) stock. It is unclear whether increased forest productivity after recovery from acidification will offset potential losses in SOC resulting from increased rates of soil microbial respiration. This study estimates the C content of three major soil types at Black Rock Forest and assesses the extent of recovery from acidification since 1995. It was hypothesized that soil type would be a major determinant of C content, and that both surface and subsoil pH would have increased across the forest between 1995 and 2019. Rock outcrop soils contain an average of 15.12 ± 3.96 kg C m^{-2}; Hollis soils 20.99 ± 5.19 kg C m^{-2}; and Swartswood soils 12.54 ± 0.59 kg C m^{-2}. Averaging the C stock of each soil type, soils account for 53% of total ecosystem C in the forest. Estimates presented here are significantly higher than in prior studies; Levy (2012) found that soils held, on average, 3 kg C m^{-2}. This study differentiates itself by incorporating methods that address the challenges of estimating C content in rocky soils. Surface soil pH significantly increased between 1995 and 2019 ($p= 3.9x10^{-5}$), while subsoils did not (p=0.529).

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Introduction

Forest Soils and the Global Carbon Budget

Forest ecosystems play an important role in the global carbon (C) cycle, with carbon sequestration and storage occurring in living biomass and in soils. There is significant variation among the soil organic carbon (SOC) stocks of various ecosystems, with soil C density generally increasing with latitude (Lal, 2005). The quantity of C stored in soils globally exceeds the quantity of carbon stored in plant biomass and the atmosphere (Scharlemann et al., 2014; Friedlingstein et al., 2019). Globally, the world's soils are estimated to collectively contain between 1,500 and 2,400 Gt of C, or about 34% of all terrestrial C, which includes oil, coal, and gas reserves, permafrost, and biomass (Friedlingstein et al., 2019). Living aboveground biomass contains 450 to 650 Gt C, approximately one third of the C stored in the world's soils (Friedlingstein et al., 2019). Specifically in temperate forests, up to two-thirds of stored carbon is held in the soils (Dixon et al., 1994).

An ecosystem's SOC stock can be altered through factors such as land-use conversion and tillage, losses of above- and below-ground biomass being returned to the soil, changes in the rate of decomposition of soil organic matter (SOM), and variations in the soil's carbon-to-nitrogen (C:N) ratio, soil mineralogy, erosion, natural disturbances, and climate (Lal, 2005; Gray and Whittier, 2014; Torn et al., 1997; Thuille et al., 1999). Of particular significance to the global carbon cycle is the impact of deforestation and land conversion to agriculture on a terrestrial carbon stocks; SOC stocks of agricultural ecosystems are diminished relative to unharvested old growth forests (Lal, 2005). Conversely forest regrowth increases a forest's capacity to sequester atmospheric carbon (Canadell and Raupach, 2008; Keith et al., 2014; Thuille et al., 1999). In decreasing order, SOC density under different land uses proceeds as

follows: forested land, orchard land, grassland, immature forest, and terraced cropland (Wang et al., 2010).

The magnitude and dynamics of a forest's SOC stock have implications for global climate, and aggrading properly managed forests may help to mitigate rising atmospheric CO_2 concentrations (Lal, 2005). Strategic conservation and restoration of terrestrial ecosystems along with improved land management together have the capacity to provide as much as 37 percent of the emissions reductions needed by 2030 to keep global warming to under 2˚C (Griscom et al., 2017). Studies informing the Trillion Tree campaign found a global potential for the restoration of 589 billion trees, which would complement government efforts to reduce C emissions (Goymer, 2018). In addition to sequestering C in living biomass, tree planting can not only mitigate diminishing soil C stocks, but can cause the soil C pool to increase. At the regional level, several forest management practices can ensure that soils remain C sinks. Improving forest productivity (by enhancing the growth potential of individual trees, for example), minimizing disturbances to stand structure, and protecting biodiversity contribute to stabilizing the soil C pool (Jandl et al., 2007). However, simply increasing the productivity of forests may not always enhance SOC stocks. Active management of soil drainage, growing species with a high net primary productivity, application of nitrogen and micronutrients, and conservation of soil and water resources can all increase the size of the soil C pool (Lal, 2005).

History of Acid Deposition and Relationship to Soil Carbon

While the impact of land-use change on soil carbon storage has been the subject of much research, the effect of acid deposition on forest soil carbon storage has been relatively less studied. Acid deposition, particularly that resulting from air pollution that peaked in the 1980s,

contributes to forest decline for certain tree species in some areas of the American northeast, with possible implications for the soil C pool (Godbold et al., 1988; Demchik and Sharpe, 2000). Godbold et al. (1988) found that aluminum leached from acidified soils accounted for root damage and tree decline. Similarly, Demchik and Sharpe (2000) found that nutrient deficiency and aluminum toxicity contributed to the decline of northern red oaks, a prominent species at Black Rock Forest. Although the connections between forest decline due to acid deposition and soil carbon storage are not clear, studies of forest decline as a result of timber harvest reveal significant reductions in the SOC stock (Evans et al., 2012; Nave et al., 2009; Canadell and Raupach, 2008; Keith et al., 2014; Thuille et al., 1999). Of particular interest in this study is the impact of recovery from acidification on SOC stocks. Between 1995 and 2017, nitrate and sulfate ion wet deposition were drastically reduced, with potentially significant effects on the soil C pool (Figure 1).

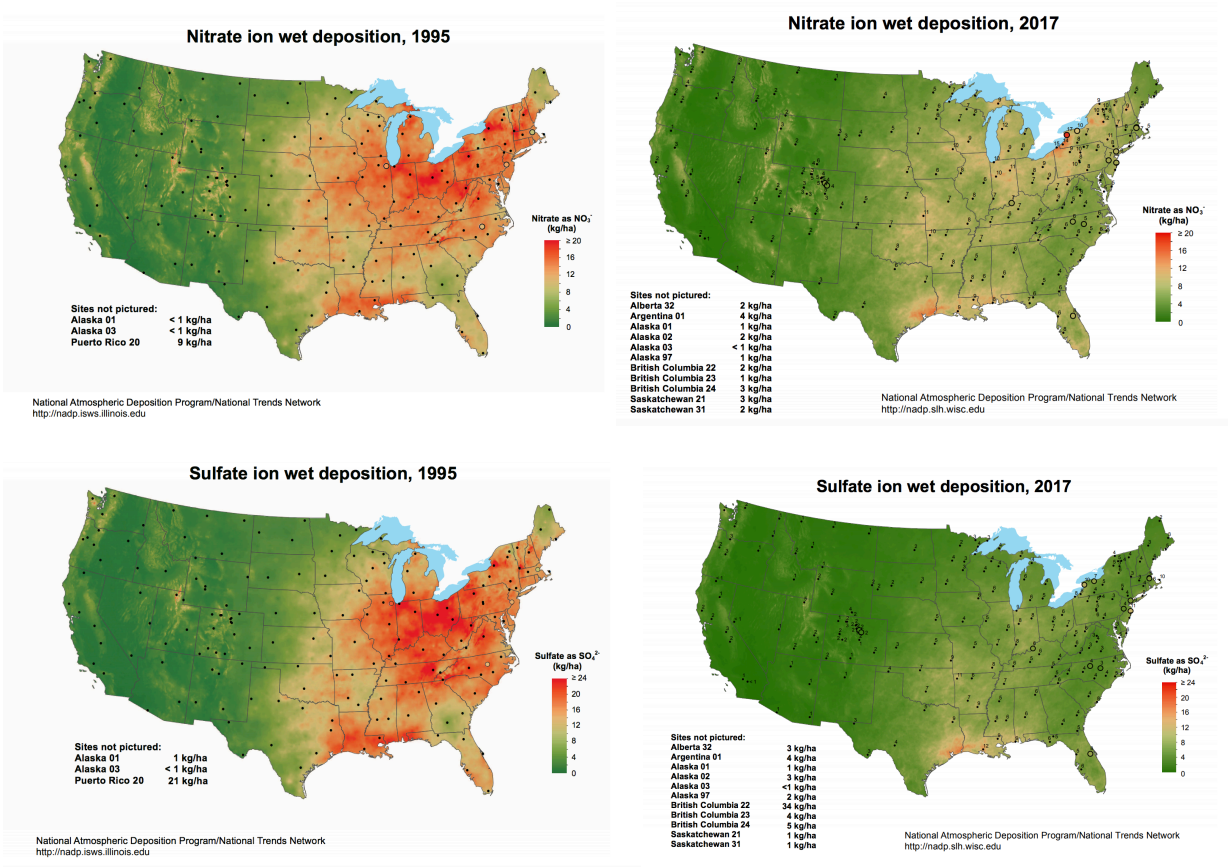


Figure 1. Changes in nitrate ion and sulfate ion wet deposition between 1995 and 2017, the most recent year for which data is publicly available. Source: National Atmospheric Deposition Program.

A study at a highly acidified spruce forest analyzed changes in soil C and nitrogen (N) pools resulting from decreased sulfur (S) deposition over a span of fifteen years (Oulehle et al., 2011). A reduction in S deposition was found to impact forest organic matter cycling, and ultimately resulted in a significant loss of accumulated SOC and N stocks (Oulehle et al., 2011). The loss of SOC and N was attributed to increased decomposition rates under less acidic conditions of the organic matter which had accumulated during a historic period of high acidification, during which decomposition was suppressed (Oulehle et al., 2011).

A study in a mixed coniferous-broadleaved forest found that acid deposition due to acid rain has a cumulative effect on soil respiration overall and heterotrophic respiration in particular,

with the rates of each declining significantly in response to higher levels of soil acidity (Meng et al., 2019; Li et al., 2019). Acid rain decreases soil pH and may introduce certain toxic ions, most notably Al^{3+}, to the soil, causing harm both to roots and soil microbial communities (Kuperman, 1996). Long-term acid deposition from acid rain can adversely impact soil decomposer activity, thus depressing the turnover rate of organic matter and increasing SOM content (Kuperman, 1996). A controlled experiment found that simulated acid rain treatments decreased soil respiration and significantly increased SOC content (Li et al., 2019). The same study also showed a significant negative correlation between soil respiration rate and SOC content. Fine root biomass decreased in response to treatment, however the net effect was an increase in the soil carbon stock (Li et al., 2019). In sum, chronic acid deposition from acid rain does affect SOC stocks, however the nature of the relationship may vary in different situations.

The 1990 amendments to the Clean Air Act are largely responsible for the improvements in rates of acid deposition (Figure 1; EPA, 2017). The key question that remains is whether increased forest productivity at recovered pH levels will offset possible losses in SOC resulting from increased rates of soil microbial respiration. In an acid mitigation experiment, Marinos and Bernhardt (2018) found that, in temperate forests, increased soil pH leads to diminished SOC stocks, with losses via respiration offsetting gains in primary productivity. The authors conducted a year-long container experiment on acid-impacted soils and found that, in the presence of plants, soil microbial respiration was significantly stimulated by an increase in pH. Demchik and Sharpe (2000) found that on study plots with high red oak mortality, both surface and subsoils had low levels of soil bases, high levels of available aluminum, and a low ratio of calcium to aluminum. Similarly, Marinos and Bernhardt (2018) found that soil Ca enrichment resulting from decreased acidification had no impact on soil microbial respiration but did

increase the growth rate of plants. Acidification reduces the calcium content of soils, a nutrient that is important to plant growth. The study by Marinos and Bernhardt (2018) suggests that, despite increased growth rates of vegetation due to Ca fertilization, losses in the SOC stock may outweigh any gains in primary productivity. The key findings of Marinos and Bernhardt (2018) have yet to be replicated in the field.

Challenges of Estimating Soil C in Rocky Soils

Managing forests for C requires accurate estimates of the terrestrial C stock. There are many sources of variability and uncertainty in attempting to assess the soil carbon pool, particularly in rocky soils (Huntington et al., 1988; Galbraith et al., 2003; Rau et al., 2011; Throop et al., 2012; Vadeboncoeur et al., 2012; Poeplau et al., 2017). Estimating SOC stocks requires good estimates of carbon content, bulk density, rock fragment content, and depth of respective soil layers (Poeplau et al., 2017). SOC stocks can be systematically overestimated if these parameters are misestimated or ignored (Poeplau et al., 2017).

Field sampling methods also have implications for the accuracy of SOC estimation. One study found that soil cores produced soil C estimates that were 16% greater than estimates produced using soil pit excavations. Perhaps relatedly, the soil cores had estimates of rock fragment mass that were 8% lower than those obtained using excavated pits (Rau et al., 2011). Soil cores give a biased representation of rock content, as they do not sample and thus don't account for the presence of larger rocks. During sample collection, several attempts may be needed to insert a core without encountering large rock fragment, leading to sampling bias for less-rocky sampling locations. While many soils are fine-textured enough to accurately estimate

bulk density and C content with soil cores, especially rocky soils necessitate alternative methods (Throop et al., 2012).

Elucidating both the dynamics of the SOC stock in northern temperate deciduous forests and the nature of the relationship between acid deposition and carbon storage is critical in a time of rapid anthropogenic climate change. Given that the organic carbon contained in soils globally constitutes approximately two-thirds of the terrestrial carbon pool, understanding the capacity of soils to store or release carbon – and the role that forest and air pollution management plays in this dynamic – is vital in mitigating the climate crisis. Warming under a business-as-usual climate scenario could result in the loss of about 55 Gt C from upper soil horizons by 2050 (Crowther, 2019). Given the level of uncertainty in estimating the magnitude and dynamics of the SOC stock, developing and implementing methods that address sampling and analytical biases is necessary, as is tracking changes in the SOC stock over time.

Thesis Statement

The purpose of this study is to estimate the SOC stock of Black Rock Forest, as well as to assess changes in soil pH since the 1990 Amendments to the Clean Air Act. Continued monitoring following this study can ultimately evaluate whether changes in soil acidity impact carbon storage. It was hypothesized that soil type would be a major determinant of C content, and that both surface and subsoil pH would have increased across the forest between 1995 and 2019. The results from this study indicate that further research is needed to understand the impacts of forest recovery from acidification on soil C sequestration and storage.

Methods

Study Site

This research was carried out at Black Rock Forest (BRF), an oak-dominated temperate deciduous forest occupying 1,550 ha in Cornwall, New York, USA. Like much of the Hudson Highlands, BRF is marked by distinct glacial geology. Till is the primary soil parent material found within the boundaries of BRF (Denny, 1938). Soils in the weathered zone are loose and more boulders may be found than in the till that lies below (Denny, 1938).

Three major soil types collectively cover 92% of BRF land area (NRCS, 2018). Rock outcrop Hollis-complex, hereafter referred to as Rock outcrop, makes up 70%. This soil type is comprised of a combination of exposed bedrock and shallow, drained Hollis soil and is generally found on hilltops and ridges. Hollis soils make up 13%, and are shallow, well- to excessively-drained, and formed in glacial till deposits. Swartswood and Mardin very stony soils, hereafter referred to as Swartswood soils, make up 9% of the forest. Swartswood soils are relatively deep and range from well-drained to moderately well-drained. They also form in glacial till deposits on hill crests, hilltops, and ridges (Table 1; Figure 2).

Table 1. Description of major soil types at Black Rock Forest as reported by the NRCS (2018).

Soil Category	**Rock Outcrop**			**Hollis**		**Swartswood**	
Type	1 (ROC)	2 (ROD)	3 (ROF)	1 (HLC)	2 (HLD)	1 (SXC)	2 (SXD)
Profile	Oa: 0-7cm H1: 7-20cm H2: 20-35cm H3: 35-55cm	Oi: 0-5cm A: 5-18cm Bw: 18-40cm 2R: 40-26cm	Oa: 0-8cm H1: 8-20cm H2: 20-28cm H3: 28-56cm	Oa: 0-7cm H1: 7-20cm H2: 20-45cm H3: 45-55cm	Oa: 0-7cm H1: 7-13cm H2: 13-38cm H3: 38-56cm	H1: 0-7cm H2: 7-78cm H3: 78-152cm	H1: 0-5cm H2: 5-71cm H3: 71-152cm
Description	Hollis interspersed with exposed bedrock	Hollis interspersed with exposed bedrock	Hollis interspersed with exposed bedrock	Gravelly loam beneath highly decomposed plant material	Gravelly loam beneath highly decomposed plant material	Gravelly loam/silt loam	Gravelly loam/fine sandy loam

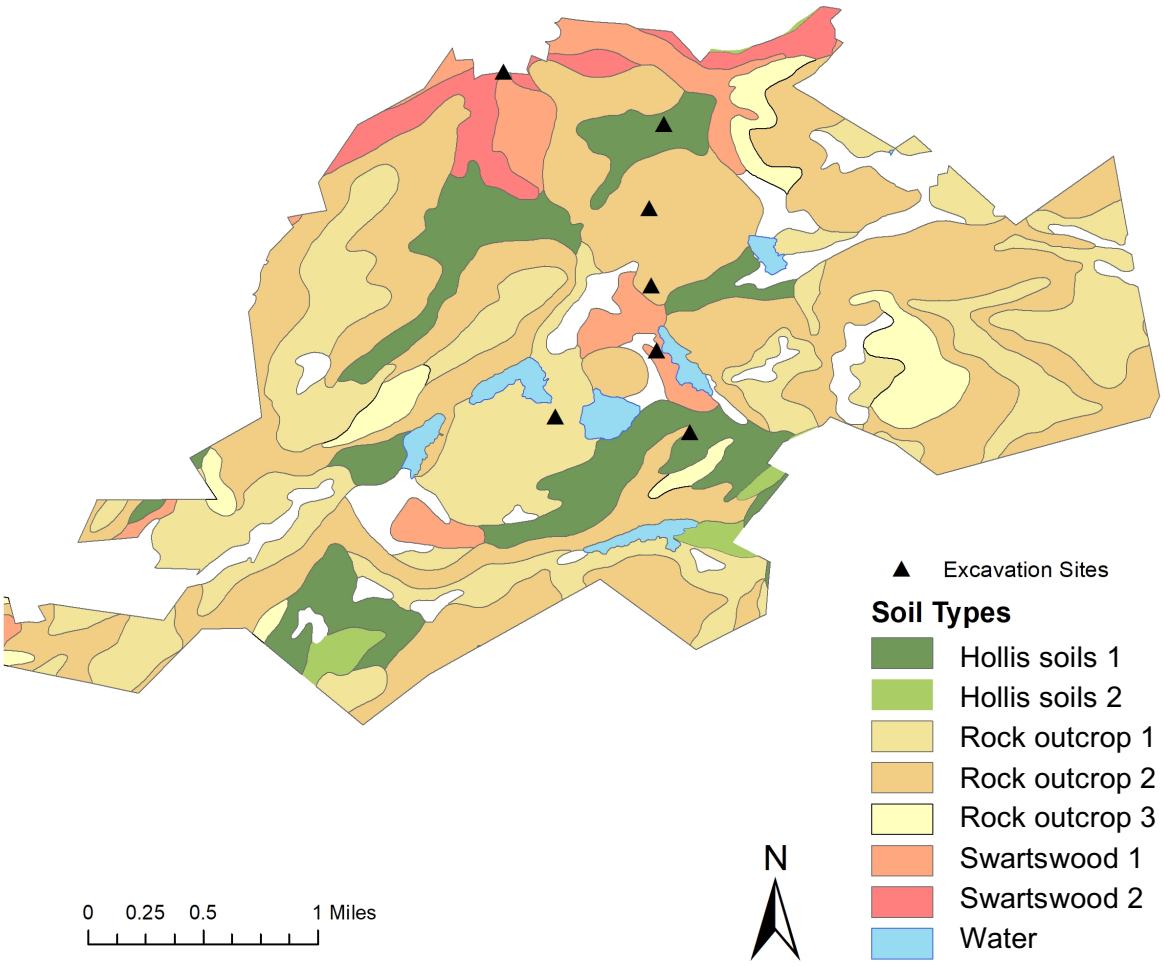


Figure 2. Major soil types of Black Rock Forest and locations of soil pits excavated. Data source: NRCS Soil Surveys, 2019. See Appendix (Figure 1) for rough overlay of Black Rock Forest trail map.

Given the especially rocky soils of BRF, broad surveys that don't account for regional particularities most likely overestimate the soil carbon pool. Bedrock and rock fragments in this region contain negligible amounts of carbon and therefore must be excluded from extrapolations made based on soil samples taken in the field. A study based on German Agricultural Soil Inventory data found that in soils with high rock content (>30%), SOC stocks can be overestimated by greater than 100% (Poeplau et al., 2017). According to a global SOC map created by the Food and Agricultural Organization and Intergovernmental Technical Panel on

Soils (2019), the carbon content of the BRF region of New York is estimated to be about 7.5 kg C m^{-2}. Rock fragment was estimated as opposed to measured directly for these estimates (FAO and ITPS, 2019). Localized study that takes into account regional geography is likely to improve estimates of the SOC stock.

The ecosystem carbon pool, and in particular aboveground wood volume and biomass, have been research topics of particular interest at BRF since the 1930s. With empirical data on live aboveground biomass, coarse woody debris, roots, and forest floor litter, the literature only lacks data on soil carbon (Schuster et al., 2008). This study seeks to remedy that, as well as add to existing data on ecosystem recovery from acidification, which thus far has primarily been investigated in the lakes and ponds of Black Rock Forest (Singh-Smith, 2019). The portion of the study concerned with pH was carried out across eight long-term study plots. To monitor changes in the forest, sixteen long-term plots were established between 1931 and 1936 (Schuster et al., 2008). The areas including these plots were clear-cut most recently between the late nineteenth century and early 20th century (Schuster et al., 2008). The thinned plots were then experimentally manipulated between 1930 and 1936, with the control plots having not been treated at all since their previous clearcutting.

Field Methods

To estimate the SOC stock, seven pits were excavated between June and September 2019. The quantitative pit method was chosen over using a soil core in order to account for the presence of rock fragments (Harrison, 1994; Rau et al., 2011; Huntington et al., 1988; Vadeboncoeur et al., 2013). Each pit was excavated to a depth of 55cm and was approximately 60cm by 30cm in length and width. However, the methods employed for the first six pits varied

somewhat from those employed for the seventh pit. Of the first six pits, two were excavated in regions where rock outcrop is the prominent soil type, two in regions of Hollis soils, and two in regions of Swartswood soils. A cuboid of soil, henceforth referred to as the secondary excavation, with a length and width of 20cm and a depth of 55cm was excavated from a cleared wall of the excavated pit. All of the soil from the secondary excavation was collected for the purpose of rock content estimation but not archived. Within the primary pit, soil horizons were identified and depths recorded. A portion of soil of unspecified volume was collected from each horizon, to be used for laboratory analysis of carbon content. At each excavation site, a sample of leaf litter from a $0.24m^2$ area was collected.

For the seventh pit the secondary excavation, also cuboidal, had a length and width of 5cm and a depth of 58cm. Prior to excavation, soil horizons were identified and the depths recorded. The excavation was carried out horizon-by-horizon, and the samples (here of known volume) were retained separate from one another and archived for analysis of carbon content.

To study the spatial variation of pH across the forest, 80 soil samples were collected from eight long-term study plots (Table 2). At each plot, a surface and subsoil sample were collected from each of the four corners and from the center of the plot, totaling ten samples per plot. A soil shovel was used to obtain a wedge of soil, which was subsequently split into the surface and subsoil portions. Each sample was approximately 0.3 liter by volume.

Table 2. Name and corresponding soil types for each long-term plot.

Plot Code	Plot Name	Soil Type
4a1C	Arthur's Brook Control	Rock outcrop
4a1	Arthur's Brook Thinned	Rock outcrop
4a2C	White Oak Control	Hollis
4a2	White Oak Thinned	Hollis
5a4C	Mount Misery Control	Rock outcrop
5a4	Mount Misery Thinned	Rock outcrop
9c1C	Bog Meadow Control	Rock outcrop
9c1	Bog Meadow Thinned	Rock outcrop

Laboratory Methods

The soil retained from the secondary excavations at the first six locations was sieved to remove all particles >0.6 cm. All coarse fragments not passing the 0.6cm filter were rinsed and the volume measured by water displacement. Soil samples taken from the horizons of the primary pit, collected separately from the soil retained from the secondary excavations, were analyzed using the loss-on-ignition (LOI) method. Samples were dried at 80˚C for 24 hours, weighed, sieved to 2mm, weighed, combusted at 550˚C for three hours, and weighed once again. The difference in mass immediately prior to and after combustion was multiplied by 0.50 to derive the carbon content of the sample (Pribyl, 2010; Jensen et al., 2018; Hoogsteen et al., 2015). The samples taken from the secondary excavation of the seventh location were also analyzed for carbon content using the LOI method. Portions of soil were set aside prior to combustion for possible future CHN analysis.

This study assumes that particles larger than 2mm and smaller than 6mm contain negligible quantities of C and do not contain a significant amount of rock. The material between 2 and 6mm was primarily small pebbles and stones. Therefore, that fraction of the soil sample

likely contains little C, however excluding those rock particles from the rock content estimation may result in an underestimation of the rock fragment percent. However, in analyzing the samples taken from the horizons of the primary pit, the mass of soil not passing the 2mm sieve was included when calculating $volume_{sample}$ (Equation a). Additionally, the material between 2 and 6mm contained some roots. An estimation of the C content of roots is not included in this study, however root biomass should be counted in estimating the belowground C stock.

Leaf litter was analyzed for carbon content using LOI. Samples were dried at 80˚C for 24 hours, weighed, then combusted at 550˚C for five hours and weighed again. The difference in mass before and after combustion was multiplied by 0.50 to determine the carbon content.

A minimum of 5.0g of soil from each of the 80 samples taken for pH analysis was retained and used to conduct individual slurry tests; samples were combined with an equivalent volume of deionized water, heavily shaken, then soil was allowed to settle and the pH of the supernatant was measured. From the remaining soil, aggregate samples were created by combining equal portions of each sample from a particular plot, with separate aggregates for surface soils and subsoils. These aggregate surface soil and subsoil samples for each long-term plot (16 aggregate samples in total) were sent to the Cornell Soil Health Testing Laboratory for analysis of pH, LOI, and nutrient analysis.

Data Analysis

The following equations, adapted from Poeplau et al. (2017) and Levy (2012) are used to estimate the SOC stock at BRF:

$$FSS_i = \frac{(1 - RC) \times mass_{fine\ soil}}{volume_{sample}} \times depth_i \quad \text{(a)}$$

$$SOC\ stock_i = SOCcon_{fine\ soil} \times FSS_i \quad \text{(b)}$$

Where RC is the rock content proportion expressed as a decimal, FSS_i is the fine soil stock of the investigated layer, $mass_{fine\ soil}$ is the mass of the fine soil (<2mm) fraction, $volume_{sample}$ is the total volume of the sample, $depth_i$ is the depth of the investigated layer, $SOCcon_{fine\ soil}$ is the SOC concentration of the fine soil layer, and SOC $stock_i$ is the SOC stock of the investigated layer. Volume and surface area are not known for samples taken from the first six excavations. To address this, equation (a) is modified by replacing $volume_{sample}$ with ($mass_{sample}$ / bulk density of the investigated soil later), with bulk density calculated with samples from the seventh excavation. The carbon content determined by these analyses was then extrapolated across the forest.

Soil pH analysis was conducted by calculating the average change in pH of each long-term plot between 1995 and 2019, and by using one- and two-sample t-tests to determine the statistical significance of observed changes.

Results

Soil Carbon Estimates

Rock outcrop soil types have the highest rock content by volume, with an average coarse fragment percentage of 20.50 ± 5.0% (Table 3). Swartswood soils have the lowest coarse fragment percentage, at 11.00 ± 2.7% (Table 3).

Table 3. Rock fragment variation between major soil types.

	Rock outcrop (n=3)	**Hollis (n=2)**	**Swartswood (n=2)**
Mean rock content by volume (%)	20.5	13.2	11.0
Standard error (%)	5.0	4.3	2.7

Consistently cross the three soil types, the majority of C contained in the soils was found in the top 10cm of the profile. Of all soil layers, the leaf litter layer contains the highest percent C, ranging from 45 to 48% C (Figure 3). Soil layers below 30cm of depth are comprised of less than 3% C (Figure 3). The relationship between percent C and depth is consistent among the three soil types (Figure 3).

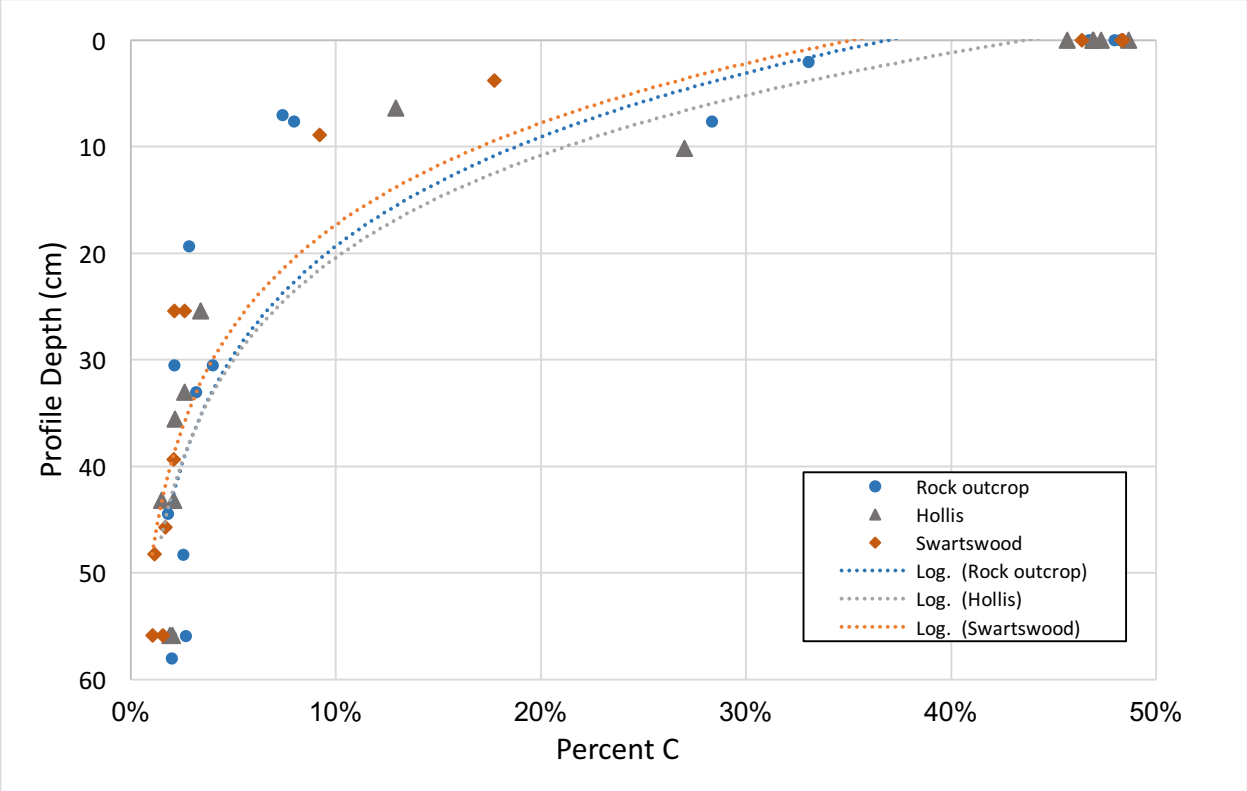


Figure 3. Percent C decreases with depth for each soil type.

After calculating carbon content following Equations a and b, Hollis soils contain, on average, the highest C density with 20.99 ± 5.18 kg C/m^2 (Table 4). Rock outcrop soils contain an average of 15.13 ± 3.96 kg C/m^2, and Swartswood contain the least C, with 12.54 ± 0.59 kg C/m^2 (Table 4).

Table 4. Organic carbon content across soil types.
* denotes seventh pit with modified field methods and data analysis

	Rock Outcrop	**Hollis**	**Swartswood**
SOC stock (kg C/m^2)	9.73 22.84 12.81*	15.80 26.17	11.94 13.13
Mean SOC stock (kg C/m^2)	15.13 ± 3.96	20.99 ± 5.18	12.54 ± 0.59

Using these carbon densities to estimate of the forest-wide SOC stock by scaling according to spatial extent of each soil type, rock outcrop soils contain a total of 176,278 Mg C, Hollis soils contain 45,550 Mg C, and Swartswood soils contain 18,839 Mg C. The total SOC stock of Black Rock Forest is 240,668 Mg C, with a weighted average of 157 Mg C/ha. This estimate accounts for the area of forest covered by ponds.

Soil pH

Soil recovery from acidification was not observed at all study locations. However, on average the increase in surface soil pH between 1995 and 2019 on all plots was significantly positive ($p=3.9x10^{-5}$). On average, subsoils across all plots did not significantly recover ($p=0.529$). The mean change in pH of surface soil is significantly different from that of the subsoil ($p=0.003$).

Increases in average pH were observed in the surface soils for seven of eight plots, and increases in the pH of the subsoils were observed in four of eight plots (Table 5). pH testing of a homogenized sample for each plot shows that the surface soil of six out of eight plots recovered, and subsoil from five of eight plots recovered (Table 5). In certain cases, the variation of change

in pH between composite and individual samples is substantial. These differences are not consistently one-direction (Table 5).

Table 5. Changes in surface soil and subsoil pH from 1995 to 2019 across all study plots. Change in pH was calculated both using the average of multiple samples, and using one homogenized sample tested at Cornell Soil Health Testing Laboratory.

Plot	Average change in pH (multiple samples)	Change in pH (homogenized sample)
Arthur's Brook Thinned		
Surface soil	0.84 ± 0.39	0.87
Subsoil	0.77 ± 0.33	0.89
Arthur's Brook Control		
Surface soil	0.62 ± 0.25	0.48
Subsoil	0.64 ± 0.19	0.68
White Oak Thinned		
Surface soil	0.20 ± 0.06	–0.03
Subsoil	0.02 ± 0.18	0.35
White Oak Control		
Surface soil	0.38 ± 0.06	0.01
Subsoil	–0.70 ± 0.03	–0.49
Mount Misery Thinned		
Surface soil	–0.10 ± 0.17	–0.20
Subsoil	–0.26 ± 0.05	–0.05
Mount Misery Control		
Surface soil	0.04 ± 0.06	0.10
Subsoil	–0.30 ± 0.09	–0.15
Bog Meadow Thinned		
Surface soil	0.26 ± 0.07	0.10
Subsoil	–0.35 ± 0.14	0.03
Bog Meadow Control		
Surface soil	0.74 ± 0.26	0.41
Subsoil	0.12 ± 0.09	0.18

There is variation in the extent of soil pH recovery between surface soils and subsoils in each study plot (Figure 4). The average change in pH of all surface soil sampled is 0.37, and for subsoil is –0.01 (Figure 4).

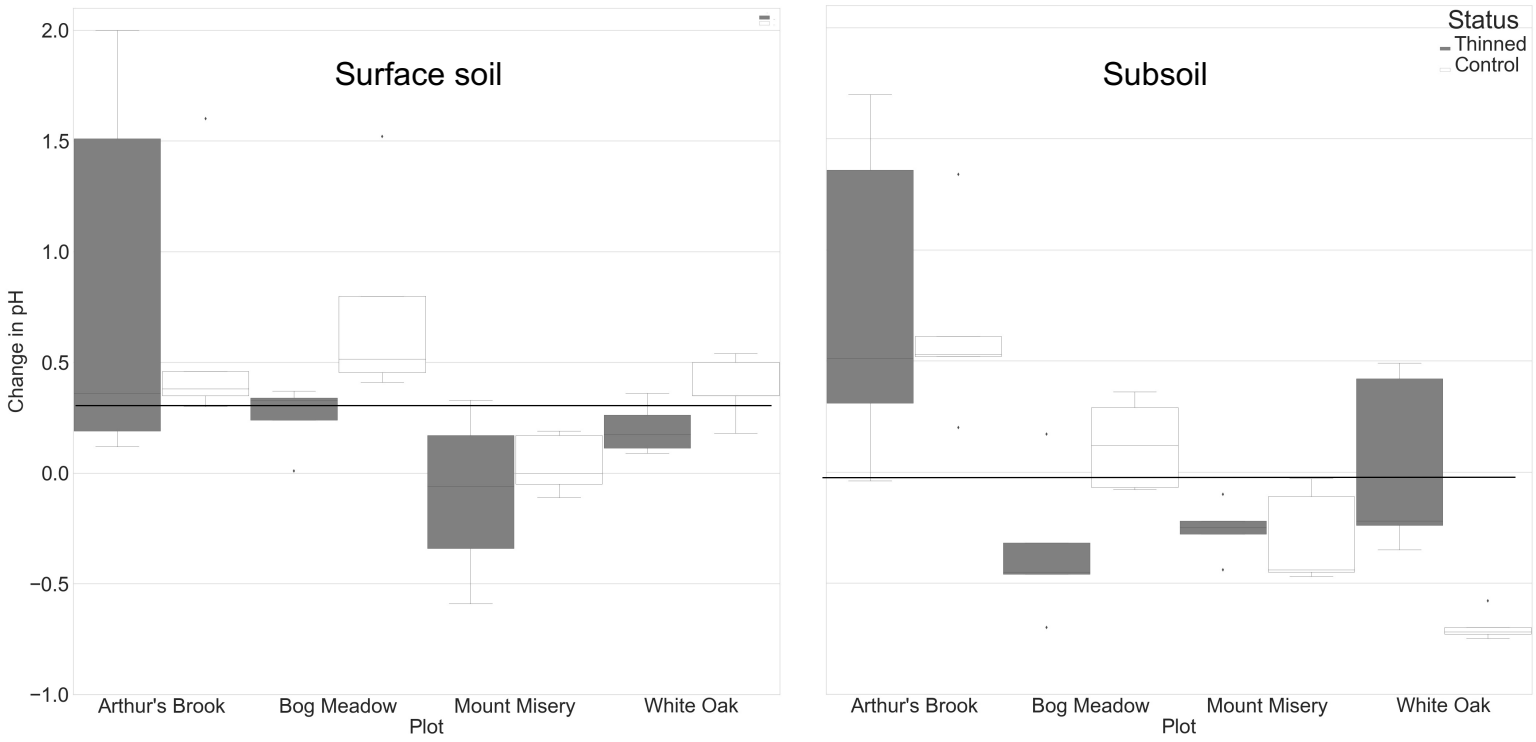


Figure 4. Change in surface soil pH at each long-term plot between 1995 and 2019. Mean change in surface soil pH, indicated by black line, is 0.37. Mean change in subsoil pH is –0.01.

Discussion

Estimating Soil Carbon

Accurate assessments of ecosystem carbon stocks are imperative for both climate modelling and forest management. This study provides an updated and expanded assessment of the soil C stock at Black Rock Forest. The only prior study to estimate the SOC stock of Black Rock Forest found that soils contained between 2.6 ± 0.5 kg C m^{-2} and 3.3 ± 0.5 kg C m^{-2} (Levy, 2012). The estimates derived here are higher, ranging from an average of 12.5 kg C m^{-2} in Swartswood soils to 21.0 kg C m^{-2} in Hollis soils. There is reason to believe that the estimates presented here more accurately resemble the actual SOC stock of the forest.

A possible explanation for the disparity in estimates lies in Levy's (2012) use of a soil core to sample. To address the bias surrounding large rock fragments, Levy (2012) estimated the

rock content of the forest's soils at 24% based on data from a single soil pit. This study found that rock content ranges from 11.00 ± 2.4% to 20.50 ± 0.05% based on sampling from seven pits across the three primary soil types. These lower estimates of rock fragment percentage result in higher estimates of soil C. To quantify the differences resulting from differing rock content estimates, Levy's (2012) rock content value was tested against the rock content values put forth in this study. The values in this study, relative to Levy (2012) result in a C estimation that is 5% higher for Rock outcrop soils, 14% higher for Hollis soils, and 17% higher for Swartswood soils (Appendix Table 1). These values indicate that difference in rock content estimation alone are not sufficient to explain the higher C density estimated here.

One other potential driver of the difference is that samples taken by Levy (2012) only reached a depth of 30cm, thus ignoring the C present at greater depths in the soil profile. While markedly lower in C than the litter layer and surface soil, which ranges from 10 to nearly 50% C, the depth of the deeper, mineral soils (B and C horizons) greatly exceeds that of the organic matter-rich surface soil (Figure 2). These findings suggest that while deeper soils are less carbon-dense than surface level soils, the quantity of C stored may be considerable due to the high volume of soil present. Averaging data for each soil type, soils from 30 to 60cm of depth hold 2.12 ± 0.16% C (Figure 3). The depth of the soil profile is highly variable throughout the forest. In one study, soil samples were taken at a depth of 122cm, and the C content was 0.38% C (Paek, 1998). Paek (1998) did not attempt to estimate the total quantity of soil C stored at BRF, however their sampling confirms that some regions within the forest are characterized by a deep soil profile. Like the rock content estimate, the amount of C held at depth does not entirely explain the difference between soil C estimates. Adding the estimate of C held below 30cm of depth only approximately doubles Levy's (2012) estimates.

A final explanation for the higher estimates in this study lies in the potential for spatial variation of the soil C pool in a forest with heterogeneous terrain. The regions sampled in this study varied in their elevation, slope, rockiness, tree canopy cover, and plant species composition. The findings of this study suggest that variation in soil C may not be solely influenced by soil type, as was hypothesized. As Don et al. (2007) note, the SOC stock depends on both the SOC concentration and the bulk density of the fine soil fraction. Each of these variables is influenced by a variety of biotic and abiotic factors (Don et al., 2007). A study on spatial variation in the soil C pool in an old-growth temperate forest found multiple determinants. The most significant determinants were soil moisture and pH, however topography and canopy composition also factored (Yuan et al., 2013). Such conditions determine the decomposition rates of organic matter, demonstrating that SOC is spatially structured. In addition, these determinants are interrelated; topography can influence variability in soil quality and characteristics (Yuan et al., 2013).

Several additional sources of uncertainty in this study remain. It is possible that the soil C estimates presented here are underestimated. The forest floor litter layer generally peaks between December and January and reaches a seasonal minimum in late summer, and therefore the average volume of these pools may be higher than is reflected here (Coleman et al., 2017). Additionally, in calculating the total SOC stock for samples lacking volume and surface area measurements, equation (a) was modified such that the variable $volume_{sample}$ was replaced by sample mass (known) over bulk density. Bulk density was calculated from samples taken at the seventh excavation site. The high variability of the soil C pool suggests a potentially high variability in soil bulk densities across the forest, and therefore applying a single bulk density estimate to soil C calculations across the forest introduces uncertainty. High C soils have a lower

bulk density, and the soil samples from which bulk density was calculated were relatively high in C. The bulk density estimate could have been too low for some of the calculations to which it was applied, resulting in possible underestimation of soil C.

Additionally, this study does not include Black Rock Forest's wetlands. One study found that on average, peatlands north of 45˚N are comprised of 49 ± 2% C (Loisel et al., 2014). Histic humaquepts soils, characterized by mucky gravelly silt lome, make up 0.7% of Black Rock Forest area (NRCS, 2018). Natchaug and Wawayanda soils, characterized by muck and silt loam, make up 0.6% of forest area. If these soil types are taken to represent the forest's wetlands, then approximately 1.3% of forest area is carbon-dense wetland. Black Rock Forest's wetlands (approximately 41.4˚N) are very likely significant C pools, however further study is required to understand their capacity for C sequestration and storage.

Two final sources of uncertainty lie in the exclusion of material in the 2-6mm fraction from analysis. That fraction holds both small stones and roots. The rock content estimation here is likely underestimated as a result of the exclusion of these small rock fragments. Roots not passing the 2mm sieve contribute to the belowground C pool, resulting in possible underestimation of the soil C pool.

It is also highly likely that the SOC stock proposed here is overestimated due to a lack of data on exposed bedrock in the forest. Based on informal observation, up to 10% of the forest area may be comprised of exposed bedrock. Future studies should utilize field methods or analyze snowless and leafless aerial images of the forest to estimate the percentage of the forest comprised of exposed bedrock and other large rock outcrops using GIS. The extent of over- or underestimation is not quantified, and therefore the net effect of all uncertainties is unknown.

Additional sampling that addresses the limitations discussed here is necessary to improve upon the estimates provided by this study.

The estimates put forth here complete Black Rock Forest's ecosystem C inventory. Averaged across soil types, the SOC stock of the forest is 188.1 Mg C/ha. According to a long-term study on biomass changes at Black Rock Forest, total live tree C (above and belowground) in 2000 was estimated at 119 Mg C/ha (Schuster et al., 2008). Coarse woody debris was estimated at 19 Mg C/ha (Schuster et al., 2008). This study estimates leaf litter across the forest (excluded from the soil C estimate above) to hold 2.85 Mg C/ha. As expected, the soil pool holds more C than any other pool. According to Schuster et al. (2008), live tree biomass holds 40%, coarse woody debris holds 6%, and leaf litter holds 1%. This study estimates that soils contain 53% of total ecosystem C. The findings of this study confirm that the soil C pool in temperate forest ecosystems is significant.

Soil Recovery from Acidification

Evidence for soil recovery from acidification is somewhat inconclusive. Soils did not, across the board, recover significantly from past acid deposition. However, between 1995 and 2019 the average change in pH of forest surface soils was positive, and for forest subsoil was near zero. A study that sampled across the northeastern US and eastern Canada found that, over intervals of 8-24 years, pH values in surface soils increased at 11 sites and decreased at three sites, while subsoil pH increased at 12 sites and decreased at three sites (Lawrence et al., 2015). Out of 27 total study sites, only five sites showed an increase in both surface soil and subsoil pH (Lawrence et al., 2015). These results, and those put forth in this study, imply considerable spatial variation in the extent of soil recovery, and suggest that the dynamics of recovery may be

different between surface soil and subsoil. Further research is required to explain the variation in the extent and timeline of recovery.

In order to monitor the relationship between recovery from acidification and the soil C pool, future studies should aim to sample for soil pH and soil C at the same locations throughout the forest. This study confirms that there is significant spatial variation in both pH and C content throughout forest soils. In addition to maintaining consistency in the methods used, future research should establish long-term study sites.

Results varied between the averaged individual slurry tests and the testing of one homogenized sample at the Cornell Soil Health Testing Laboratory. This introduces uncertainty into the soil pH aspect of this study, as it is unclear which test results are closest to the true values of soil pH change. However, both sets of data, as well as estimates of the SOC stock, set a baseline from which to track changes in each.

Forest Management and the Carbon Market

As a result of this study, Black Rock Forest has the option to pursue participation in the carbon trading market. Forest management is an important component of meeting CO_2 stabilization policies that will mitigate the worst effects of climate change (Tavoni et al., 2007). Tying forest management to the carbon market could reduce the burden of policy to meet emission reduction targets by 50 ppmv, the equivalent of 0.25˚ C (Tavoni et al., 2007). Projects that sequester or offset greenhouse gas emissions, such as forest management for carbon storage, generate credits that can be sold to regulated entities or organizations intending to reduce their own carbon output (Gorte and Ramseur, 2008).

As an example, organizations like The Nature Conservancy offer financial incentives to individuals and organizations that own significant amounts of forestland, provided that the land

is conserved and managed in accordance with a set of guidelines (The Nature Conservancy, n.d.). Partaking in such programs requires completion of a full carbon budget for the forested land. Once the budget is determined, a forest's management has the option of selling carbon credits to firms wishing to reduce their carbon footprint (The Nature Conservancy, n.d.). This can serve as additional revenue for non-profit environmental and educational organizations such as Black Rock Forest.

In 2019, the Albany Water Board signed on to the Working Woodlands program, which is projected to earn the organization a million dollars over the next decade (Altamont Enterprise, 2019). The city of Albany will protect 2,590 ha of forest, much of which surrounds regions of wetland important to the city's water supply. The city has created a management plan to improve C sequestration, and is selling the carbon credits (Altamont Enterprise, 2019). Managing forests for C storage is one of the most efficient and inexpensive ways to mitigate climate change, and in addition can provide revenue to agencies and organizations who participate who sell carbon credits.

Conclusions

This research provides an estimate of the SOC stock of Black Rock Forest, thus completing an inventory of total ecosystem carbon. This study also identifies the ways in which further research can expand, particularly on the dynamics of the soil carbon pool as the forest continues to experience recovery from high rates of acid deposition. Black Rock Forest may also choose to use the data included in this study to pursue participation in the carbon trading market.

Additionally, data on soil pH across the eight long-term plots indicate some degree of recovery from historical acidification. Further analysis will elucidate the extent to which it can

be said that the 1990 Amendments to the Clean Air Act have had a positive effect on forest soil health. Finally, establishing datasets for soil pH and the SOC stock will allow for continued data collection, making future studies on the relationship between soil acidification and carbon storage possible. That recovery from acid deposition may ultimately cause organic C content to decrease intensifies the need for research on the complexities of this process.

Recommendations

Further research is needed to understand the specific causes of variability in the soil C pool at Black Rock Forest. In addition, sampling at additional sites could improve the accuracy of the soil C estimate presented here. Monitoring soil C and pH over time is recommended, as it will elucidate the relationship between fluxes in the soil C pool and the process of recovery from acidification. In future sampling of soil C, samples of a known volume must be taken in order for data to be analyzed with Equations A and B as written. Exposed bedrock should be estimated using either GIS or field tools such as a quadrat. Finally, a comprehensive assessment of the C storage in roots should be carried out.

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Appendix

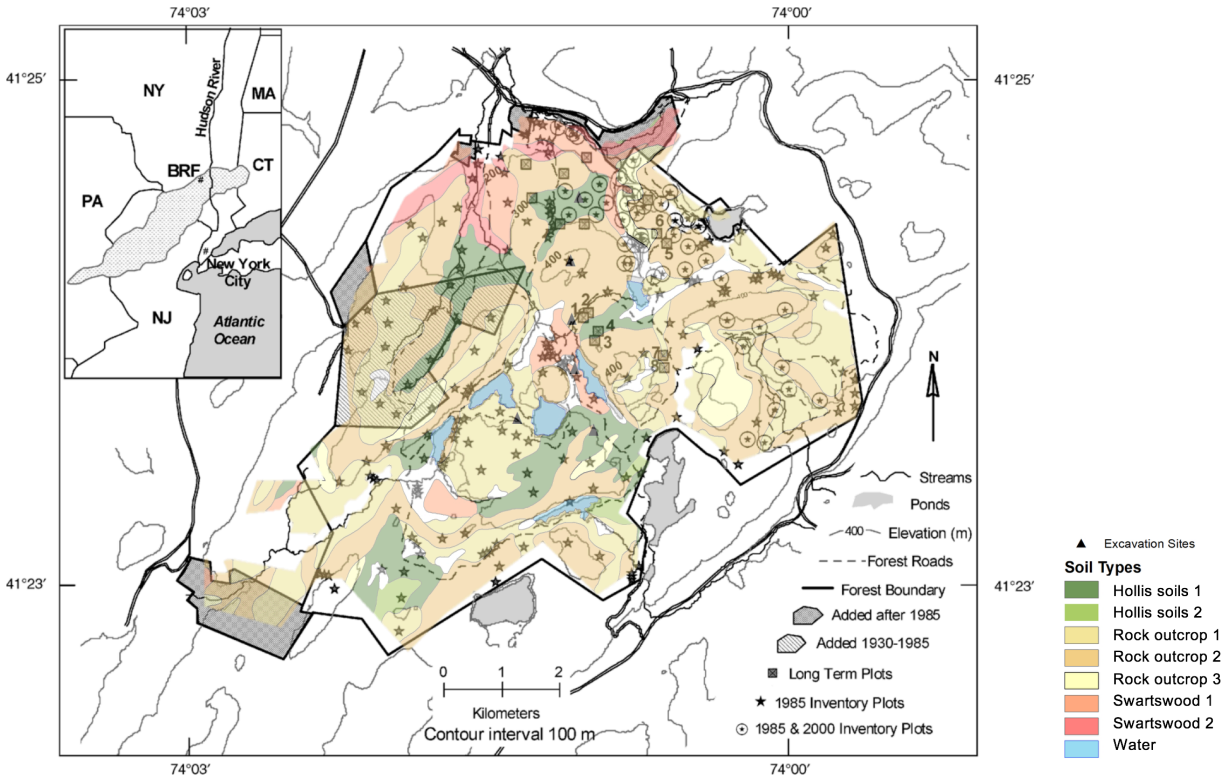


Figure 1. Soil types map roughly overlain on Black Rock Forest map (source: Schuster et al., 2008).

Table 1. Percent increase in soil C estimates from using estimates of rock content put forth here estimates those put forth by Levy (2012).

Soil Type	Rock Outcrop	Hollis	Swartswood
Percent change in soil C estimates	4.61%	14.22%	17.11%