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Abstract

Landscape carbon storage is a key component of climate change mitigation. Tree planting on degraded land has been identified as an effective carbon capture strategy, but it is unclear how various associated restoration treatments influence carbon sequestration. In this study, we measured carbon pools in jack pine (*Pinus banksiana* Lamb.) stands subject to different restoration treatments within severely damaged upland areas in the metal mining region of Sudbury, Ontario, Canada. Treatments included: i) soil amendment (liming, fertilizing and grass/legume seeding) followed by jack pine planting, ii) jack pine planting alone, iii) soil amendment only, and iv) untreated plots with some natural regeneration as a result of improving air quality in the region. Twenty-four years after the initial treatments, we measured the carbon storage in coarse and fine woody debris, herbs, shrubs, forest floor (LFH), mineral soil, and trees. The carbon pool in planted trees was 1.5 times higher in plots when soil amendments were applied. This increase in the tree carbon pool was the result of greater carbon sequestration per individual jack pine tree rather than changes in tree density or arrival of new species. Similarly, naturally regenerated white birch (*Betula papyrifera* Marshall) in the amended plots stored 1.7 times more carbon than white birch in the untreated plots. Tree planting proved essential on this landscape with natural tree regeneration rates very low even 50 years after local pollution was massively reduced. However, tree planting surprisingly did not significantly affect total carbon storage in these exposed upland sites. Total ecosystem carbon varied from 21.4 Mg ha^{-1} to 124.5 Mg ha^{-1} across all plots (mean of 67.6 ± 4.8 Mg ha^{-1}) but no statistically significant differences were observed among the treatments. Soil organic carbon (SOC) proved to be the largest carbon pool across plots and stored on average 66% of total ecosystem carbon. However, at this early stage the restoration treatments had no influence on SOC pools on this highly degraded

landscape, which still appears to be severely affected by ongoing erosion of surface soil and metal contamination. The restoration treatment also did not significantly affect the understory or downed woody debris carbon pools. This study demonstrates that the current restoration treatments used in Sudbury do influence the amount of carbon sequestrated mainly through tree restoration on this industrially degraded landscape, but long-term storage potential is still limited by poor soil conditions.

1. Introduction

Reforestation has been proposed as a means of storing large amounts of atmospheric carbon that will help to keep global temperature increase below 1.5°C by 2050, when combined with massive reductions in the use of fossil fuels (Pan et al. 2011; Griscom et al., 2017; Lewis et al. 2019). Forests primarily sequester carbon in aboveground biomass and soil organic carbon (Lal, 2005; Luyssaert et al., 2008) and are estimated to sequester approximately 20% (2 Pg yr^{-1}) of carbon emitted from fossil fuels and industry (Pan et al., 2011; Le Quéré et al., 2018; Pugh et al., 2019). Returning trees to the landscape can achieve multiple additional goals, such as biodiversity conservation, socioeconomic benefits, and food security (Chazdon & Brancalion, 2019). These benefits have led to global efforts, such as the Bonn Challenge that has the goal to restore 350 million ha of forest land by 2030 (IUCN, 2020). While reaching this goal would be a considerable achievement, the IPCC estimates that 1 billion ha of forest is required to mitigate climate change (Rogelj et al., 2018).

Attempts to identify available land for reforestation (e.g., Bastin et al. 2019) have been controversial, as several authors have noted that identified land is located in areas where tree planting is unsuitable, such as within fragile grassland ecosystems, at northern latitudes thus increasing albedo, or land required for agriculture, among other concerns (Delzeit et al., 2019; Grainger et al., 2019; Veldman et al., 2019). Moreover, global estimates of carbon sequestration potential of replanted forests can vary by 10-fold (Griscom et al., 2017; Fuss et al., 2018; Bastin et al., 2019) depending on various assumptions (Delzeit et al., 2019; Skidmore et al., 2019; Veldman et al., 2019), and the uncertainty of the rate of carbon uptake and storage under various types of reforestation efforts (Holl & Brancalion, 2020).

In contrast there is relatively little controversy in reforestation of severely degraded lands, although they may be subject to complex land rights (Friedlingstein et al., 2019). Globally, there is an estimated 1.7 to 1.8 billion ha of degraded land, including vast areas of mining related barrens (Kozlov & Zvereva, 2007, 2015), that could be part of an effective carbon capture strategy (Bastin et al., 2019; Minnemeyer et al., 2011). Indeed, restoration of degraded land may offer the largest carbon sequestration potential as carbon pools are generally depleted in comparison to natural ecosystems (Lal et al. 2015).

The large industrial barrens surrounding the copper and nickel smelters in Sudbury, Ontario, Canada is an internationally recognized example of extreme loss of vegetation, surface soil, and carbon storage (Winterhalder, 1995). Such large and disturbed areas also provide very important opportunities to experiment and explore various aspects of disturbance ecology and natural succession as it relates to carbon capture. We know from the literature that forests experience significant carbon loss through disturbances, such as fire or insect infestations (DeLuca & Boisvenue, 2012), but they can become effective carbon sinks relatively quickly during the revegetation period (Amiro et al., 2010).

In the 1970s, Sudbury's landscape was described as one of the most ecologically disturbed sites in Canada with approximately 17,400 ha of near complete barrens surrounding the smelters and an additional semi-barren area of ca. 64,000 ha extending further outwards (Winterhalder, 1995). Exposed hills in particular were barren, while low-lying valleys were more protected and maintained some of the original vegetation and deeper soils (Winterhalder, 1995; Kozlov & Zvereva, 2007). Massive reductions in acid and metal emissions began in 1972 at a time when Sudbury's smelters were the largest point source of SO_2 in the world (> 2 MT year^{-1}; Hutchinson and Whitby, 1977). The high emissions of SO_2 and particulate metals that were discharged from

Sudbury's smelters were phytotoxic and acidified the surrounding soils (Hutchinson & Whitby, 1977). Subsequently one smelter (Coniston) was closed and a smokestack (403 m) was erected to diffuse the remaining emissions (Hutchinson & Whitby, 1977), ultimately leading to > 95% reduction in SO_2 (and similar reductions in metal particulates) by 2020 (Hall et al., 2020). Emission reductions facilitated some natural re-growth of mainly white birch (*Betula papyrifera* Marshall) but soil pH remained low (2.0–4.5), so dolomitic limestone (calcium and magnesium) was used as a soil amendment when a large-scale land reclamation (reforestation) program began in 1978 (Beckett & Negusanti, 1990). A fertilizer and seed mixture was also applied as part of the re-greening strategy (Winterhalder, 1983). Between 1978 and 2019 the program limed 3,484 ha, fertilized 3,258 ha, seeded 3,185 ha, and planted 9,858,424 trees and 431,342 understory trees and shrubs at a total cost thus far of $33.5 million (City of Greater Sudbury, 2019). In Sudbury to date, > 3,500 ha of barrens have been treated with both soil amendments and tree planting, while a much larger area has been treated with tree planting alone (ca. 16,500 ha; pers. comm. P. Beckett).

Recently, Preston et al. (2020) demonstrated that tree growth and carbon sequestration rates on restored sites in Sudbury, Ontario were comparable to silvicultural plantations in a similar climatic region; suggesting that reforestation of industrially damaged landscapes could be part of an effective carbon capture strategy. However, the authors examined a chronosequence consisting of a single restoration treatment (soil amendment followed by tree planting) located in areas where soil erosion was less severe and site conditions were more favourable. Sudbury's landscape restoration programs consist of various treatments and attempt to treat highly exposed upland sites with much poorer site conditions (i.e., prone to drought, wind damage, erosion, temperature extremes). Building on Preston et al., (2020), our study was designed to address how different restoration treatments (dolomitic limestone, high-phosphorus fertilizer, grass and legume seed mixture, and tree-planting) in acid and metal impacted landscapes affected carbon storage particularly in challenging upland sites. We aimed to determine how different restoration techniques used to reforest highly disturbed landscapes might affect carbon storage and consequently climate change mitigation potential. To achieve this, we measured carbon pools in four different restoration treatments across a single upland watershed located near the Coniston smelter in Sudbury, a smelter that was decommissioned in 1972. We hypothesized that carbon pools would be larger (i.e., greater plant growth and inputs to the soil) with the greatest treatment effort (limed, fertilized, grass and legume seeded, and tree planted).

2. Methods

2.1. Study area landscape and climate

This study was conducted in the Daisy Lake watershed (46°27'43.46"N, 80°52'48.79"W) in Sudbury, Ontario, Canada located approximately 3.5 km southwest of the abandoned Coniston nickel and copper smelter (Szkokan-Emilson et al., 2011; Figure 1). The Sudbury region consists of undulating topography with rock outcrops and generally thin soils due to past glacial activity and large-scale erosion that occurred when the landscape was barren (Pearson & Pitblado, 1995). The study was confined to one south-facing sloping area of the watershed to minimize the heterogeneity among study plots in terms of local climate and soil contamination. Low valleys adjacent to Daisy Lake itself were a refugia for soil and vegetation during past disturbances, while the upland areas are dominated by bedrock outcrops with pockets of soil between bedrock (Supplemental Figure 1). This hilly upland terrain consists of convex crests and concave dips that were subject to soil transport down-slope so that chasms received soil deposition while crests were severely eroded (i.e., the loss of > 1 m of soil; pers. comm. P. Beckett) and sparsely vegetated (Supplemental Figure 1). We confined our study plots to upland areas that were mostly barren prior to restoration (avoiding low valleys with vegetation refugia) as this was a dominant landscape feature and to further limit plot heterogeneity. The soils in this area are generally in the Podzol and Gleysol orders based on the Canadian System of Soil Classification with Regosol order in areas that received soil deposition (Hazlett et al., 1984). The soil in the Daisy Lake watershed has a silt loam texture (high silt and sand fractions; pers. comm. P. Levasseur) with horizons consisting of a 0-3 cm forest floor (LFH) layer, 0–2 cm Ah, 2–7 cm Ae, 7–18 cm Bf, 18– >30 cm Bm/BC (CEM Environmental Monitoring, 2004). Relic forest fire ash deposits and buried organic layers are found in some mineral soil as an artifact of soil movement (Hazlett et al., 1984). The colour of the Ae layer in the study area is 10YR6/, Bf layer is 10YR3/6, and Bm or BC is 10YR5/6 (CEM Environmental Monitoring, 2004). The climate of the study area is humid continental with mild summers and cold winters, according to the Köppen Climatic Classification System (Christopherson & Byrne, 2009). Average temperatures range from -13 °C in January to 19 °C in July and average annual precipitation is 903.3 mm (Environment and Climate Change Canada, 2019).

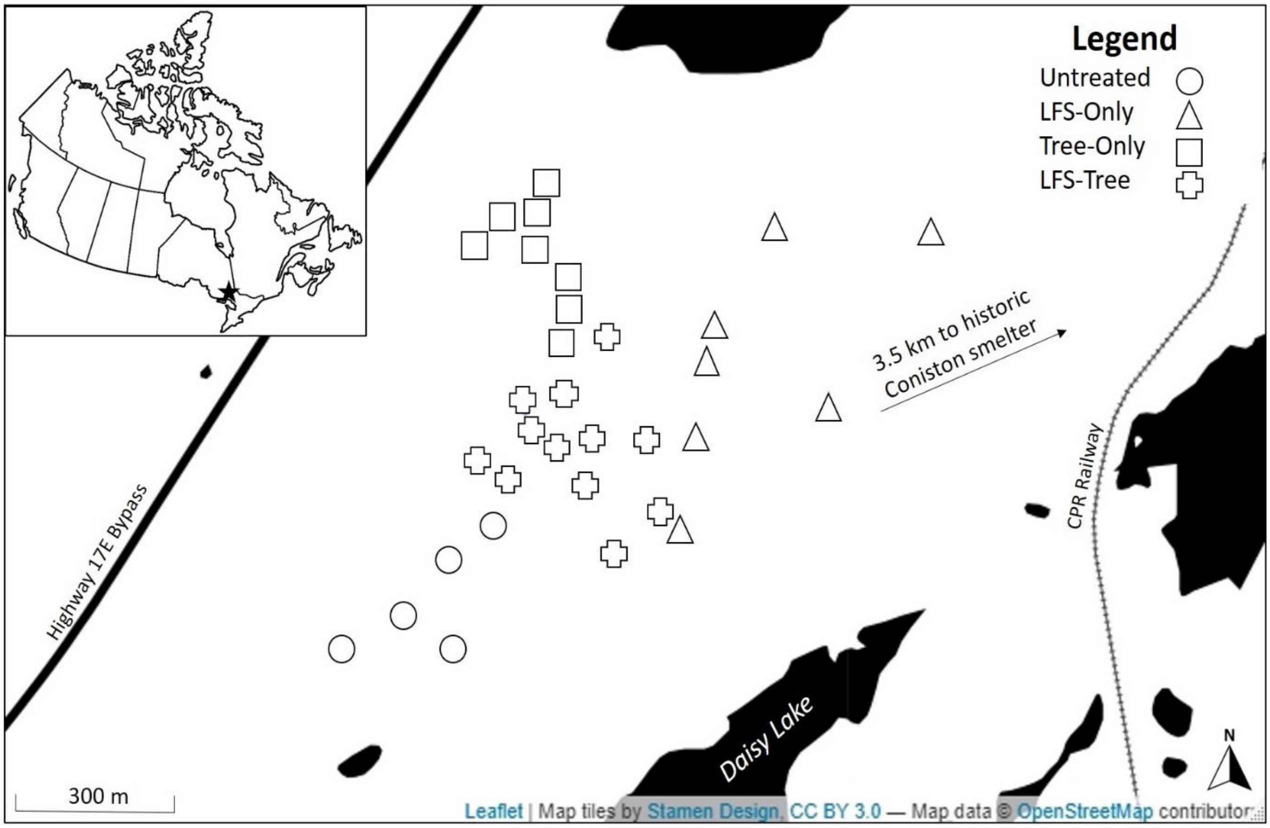


Figure 1. Location of study plots and restoration treatments within the Daisy Lake watershed, south west of the historic Coniston smelter in Sudbury, Ontario, Canada. The inset map shows the city's location within Canada.

2.2. Restoration treatments

In 1994, 38 ha of the Daisy Lake watershed were aerially limed with 10 tonnes ha^{-1} of coarse dolomitic limestone (53.9% $CaCO_3$, 44.8% $MgCO_3$; Gunn et al., 2001). Three years later (1997), a high phosphorus fertilizer (6N-24P-24K) was applied to the entire limed area by hand at a rate of 390 kg ha^{-1} and the area was then sown with a seed mixture (45 kg ha^{-1}) of grasses (10% *Agrostis stolonifera* L., 15% *Festuca rubra* L., 20% *Phalaris angusta* Nees ex Trin, 15% *Poa compressa* L., 15% *P. pratensis* L.) and nitrogen-fixing legumes (15% *Lotus corniculatus* L., 10% *Trifolium hybridum* L.; City of Greater Sudbury, 2020). Also in 1997, 1–2 year old jack pine seedlings that met forestry standards were obtained from commercial growers and planted by municipal tree planting crews in areas of the watershed that were limed, fertilized, and grass and legume seeded as well as in some un-amended areas; for a total tree planting area of 54.56 ha (City of Greater Sudbury, 2020). Restoration of the Sudbury landscape is an ongoing process and a 1.55 ha area of the watershed was infilled with additional jack pine seedlings in 2001 (City of Greater Sudbury, 2020). Jack pine was planted at a density of ca. 1,200 stems ha^{-1} but tree placement was haphazard (i.e., not in lines) to recreate natural conditions while seeking the best possible site conditions (i.e., depth of soil).

2.3. Plot selection within each treatment

A total of 32 circular plots (0.1 ha) were established in 2018 (24 years after initial restoration) to assess conditions under four different restoration treatments at the Daisy Lake watershed that included: i) limed, fertilized, seeded, and jack pine planted (LFS-Tree; n = 12); ii) limed, fertilized, and seeded (LFS-Only; n = 7); iii) jack pine planted in untreated soil (Tree-Only; n = 8); and iv) untreated soil with no restoration treatment (Untreated; n = 5; Figure 1; Figure 2). More plots and a balanced sample design were originally planned with plots located in both upland barrens and lowland valleys. However, during sampling it became clear that the lowland plots were vegetated prior to Sudbury's regreening efforts due to the protective effect of the valleys from smelter fumigation. Consequently, incorporating the lowland plots in our study would not have facilitated calculating the amount of carbon sequestered as a result of the restoration treatments themselves. Therefore, the lowland plots were excluded from our analysis. Ecosystem carbon pools within each plot were measured in the spring and summer of 2018.

Figure 2. Photos of the restoration treatments: (A) Untreated, (B) LFS-Only (limed, fertilized, grass and legume seeded), (C) Tree-Only (jack pine planted), and (D) LFS-Tree (limed, fertilized, grass and legume seeded, and jack pine planted).

2.4. Carbon pool calculation

2.4.1. Tree carbon

Within each plot all living and dead trees were identified and their health status (living or decay class 1–5; Harmon et al., 2011) and diameter at breast height (DBH; height of DBH measurement = 1.3 m) were recorded. Trees with a DBH < 3 cm were excluded from the assessment. In the absence of Sudbury-specific allometric equations, tree biomass was estimated using allometric equations derived from Canada-wide data (Lambert et al., 2005; Ung et al., 2008). Specifically, equations from Lambert et al. (2005) were used to estimate the biomass of red maple (*Acer rubrum* L.), jack pine, red pine (*Pinus resinosa* Aiton), eastern white pine (*P. strobus* L.), balsam poplar (*Populus balsamifera* L.), largetooth aspen (*P. grandidentata* Michx), red oak (*Quercus rubra* L.) and eastern hemlock (*Tsuga canadensis* (L) Carrière). Whereas, equations from Ung et al. (2008) were used to estimate the biomass of white birch, white spruce (*Picea glauca* (Moench) Voss), trembling aspen (*Populus tremuloides* Michx) as well as unknown deciduous trees that were too decayed to identify (0.03% of trees). Cherry (*Prunus* spp.) and willow (*Salix* spp.) trees were not identified to the species level and only represented 0.3% of total trees identified, therefore their biomass was estimated using the unknown deciduous tree equations (Ung et al., 2008). Total tree biomass was calculated by multiplying aboveground biomass (W_A) by 1.2 (to account for belowground tree biomass; Preston et al., 2020). Carbon concentration of each tree species followed those reported for stem wood by Lamlom and Savidge (2003). Total carbon estimated per tree was converted to Mg ha^{-1} by multiplying carbon per tree by the tree species density per plot. In the case of dead standing trees, tree carbon was multiplied by a density reduction factor (Harmon et al., 2011) and structural loss adjustment factor (Domke et al., 2011) to account for reduced density and volume due to decomposition.

Tree species richness was calculated by counting the total number of tree species in each plot and finding a mean within each treatment.

2.4.2. Shrub and herb carbon

Understory and ground vegetation were sampled in four 1 m^2 quadrats per plot. Shrubs and herbs were harvested at their base and subsamples were oven-dried at 70°C for 48 h and their dry mass recorded. As per Johnston et al. (1996), total shrub and herb biomass were assumed to be 2.5 and 3 times that of the aboveground biomass, respectively, and the carbon concentrations were assumed to be 0.48 and 0.45 of the biomass, respectively (Vogel & Gower, 1998).

2.4.3. Coarse woody debris carbon

Coarse woody debris (CWD) was defined as any downed wood with a midpoint diameter ≥ 10 cm. All CWD pieces were surveyed within each plot for length, decay class, and dimensions. A five-level decay class system was used to differentiate decomposition and derive densities (Harmon et al., 2011). The volume (*V*) of each piece of CWD was calculated using the conic-paraboloid equation from Fraver et al. (2007):

$$V = \frac{L}{12}(5A_b + 5A_u + 2\sqrt{A_b A_u})$$

where L is length of CWD, A_b is the area of base end of CWD, and A_u is the area of the upper end of CWD. We then calculated CWD carbon by multiplying volume by density (Harmon et al., 2011) and a carbon ratio of 0.50 (Russell et al., 2015).

2.4.4. Fine woody debris carbon

Fine woody debris (FWD; diameter of 1.0–9.9 cm) was surveyed using line intersect sampling (LIS) along four transects that ran north-south and east-west through the centre point of each plot. The FWD was tallied based on size and decay class and FWD biomass *(W)* was calculated per size and decay class for each transect area with an equation from Van Wagner (1968):

$$W = \frac{\pi^2 S \sum d^2}{8L}$$

where S is the decay-class density based on Harmon et al. (2011), d is diameter of FWD, and L is LIS length. FWD biomass for each transect was then found by summing all size and decay class values. FWD carbon was calculated using a 0.50 biomass to carbon ratio and scaled up to plot level (Russell et al., 2015).

2.4.5. Forest floor (LFH) carbon

Forest floor (LFH) was sampled using a soil corer with a 5 cm radius at eight points per plot along two transects. Forest floor (LFH) samples were oven-dried at 60˚C for 16 hours and the dry-mass calculated. Samples were homogenized and passed through a 2 mm sieve. Organic matter content was determined through the loss on ignition (LOI) method using a muffle furnace (Pyradia, Montreal, Canada) at 550°C for 5.5 hours following Nelson and Sommers (1996) and the carbon concentration was assumed to be 50% of the ash-corrected sample dry mass.

2.4.6. Mineral soil carbon

Mineral soil was sampled along two perpendicular transects within each plot (n = 8) using a soil corer with a 1.25 cm diameter to a maximum depth of 20 cm (the length of the corer; bedrock often impeded the full 20 cm from being sampled). Mineral soil cores were divided into

0–5 cm, 5–10 cm, and 10–20 cm deep segments and transferred to a plastic sample bag to determine bulk density and carbon content. When bedrock was encountered at depths < 20 cm depth then partial depth segments were collected (i.e., if mineral soil was 15 cm deep than a partial 10–20 cm sample was collected, and it was noted that only 5 cm of soil was included). Soil depth to the bedrock within the study plots is highly heterogeneous (see Section 2.1). To avoid overestimating the SOC on the landscape additional soil depth measurements were recorded along the same transects (n = 16) using a metal rebar (50 cm in length) and the average soil depth per plot was calculated.

In the laboratory, samples were weighed, oven dried to a constant mass at 60°C for 16 hours and then passed through a 2 mm sieve to remove coarse fragments. The mass of the coarse fragments was recorded. Total bulk density was calculated by dividing the oven-dry mass by sample volume. Fine soil (< 2 mm) bulk density and volumetric coarse fragments content were calculated following Page-Dumroese et al. (1999). Organic matter content was determined through LOI, as outlined in Section 2.4.5.

We used a Sudbury-specific ash-corrected soil organic matter to carbon ratio of 47.73% to calculate soil organic carbon (SOC; mg g^{-1}; Sherman, 2005). SOC on the landscape was calculated following Ellert et al. (2008):

$$SOC = \sum_{1}^{n} D_{cs} C_{cs} L_{cs} \times 0.1$$

Where *SOC* is the SOC pool to the mean sampled depth within a plot (Mg ha^{-1}), D_{CS} is the mean fine soil bulk density of the core segment (g cm^{-3}), *Ccs* is the mean plot SOC concentration of the core segment (mg g^{-1}), and L_{CS} is the mean length of the core segment (cm) within each plot.

To avoid overestimating SOC pool by assuming the entire core consists of fine soil, the volume of coarse fragments was first removed from the total core volume (Federer et al., 1993). The corrected core length (i.e., minus coarse fragments > 2 mm) was back calculated from the corrected core volume.

Ellert et al. (2008) reported that comparisons of SOC across plots and treatments are subject to bias if there a slight difference in bulk density. Bulk density measurements are heterogeneous on Sudbury's highly disturbed and industrially impacted landscape. As such, changes in mineral soil carbon among treatments and with depth are described using soil carbon concentration (mg g^{-1}: Lee et al., 2009), which is not influenced by bulk density measurements.

2.5. Lab analysis

Soil pH was analyzed for the 0–5 cm mineral soil layer using a Fisher Accumet model AB150 pH meter (Pittsburgh, USA) with a 2:1 ratio of milli-Q to soil. Inductively coupled plasma mass spectrometry (ICP-MS) was used to measure total and bioavailable elemental concentrations in the 0–5 cm mineral soil layer. Samples were processed at the Elliott Lake Research Field Station laboratory at Laurentian University using a multi-acid (HF/HCl/HNO_3) 90°C open-block digest for total element concentrations and a weak (0.01M) $LiNO_3$ extraction procedure for bioavailable element concentrations. Arsenic, cadmium, cobalt, copper, lead, nickel, and selenium were of particular interest, as they were identified in the Sudbury Soils Study as chemicals of concern for the Sudbury region (SARA Group, 2009). Metal concentrations below detection limits were set at half of detection limit (Clarke, 1998).

2.6. Data analysis

Data analyses were performed in R 3.6.1 (R Development Core Team, 2019) using base R as well as dplyr (Wickham et al., 2018) and ggplot2 (Wickham, 2016) packages. Data were confirmed to follow normality and heteroscedasticity assumptions prior to running statistical analyses using diagnostic plots as well as Shapiro-Wilk test for normality and Levene's Test for equality of variance (performed using the FSA package; Ogle et al., 2020). All tests were performed at a significance level of 0.05. Welch's t-tests were used to compare differences in tree planting density between jack pine planted treatments. One-way analysis of variance (ANOVA) followed by Tukey's post hoc test were used to determine differences in carbon pools, stand characteristics, soil properties, and tree densities among treatments. SOC concentration (mg g^{-1}) data were natural log transformed to meet the ANOVA assumptions. When transformed data did not meet ANOVA assumptions, Kruskal-Wallis rank sum test was applied followed by Dunn's post-hoc analysis with Benjamini-Hochberg correction factor. Correlations between measured variables (e.g., soil metal with SOC and tree carbon) were explored using Spearman's rank correlation as variances were not equal.

3. Results

3.1. Site characteristics

Median stem density was significantly higher in the LFS-Tree plots (1400 stems ha^{-1}, IQR 470) compared with the LFS-Only plots (370 stems ha^{-1}, IQR 390) and Untreated plots (80 stems ha^{-1}, IQR 100; $\chi^2_3 = 17.28$, $p < 0.001$; Dunn's $p < 0.05$; Supplemental Table 1 includes detailed values for all statistical tests). Stem density was not significantly different between the LFS-Tree and Tree-Only (1115 stems ha^{-1}, IQR 235) treatments. Tree-Only stem density was significantly greater than the Untreated stem density (Dunn's $p < 0.05$). Due to the high variability in stem density observed in the LFS-Only plots, no differences were observed between this treatment and Tree-Only or Untreated treatments.

A total of 13 species were identified across the different treatments (and one unidentified dead tree in Tree-Only), and all 13 identified species were recorded in the LFS-Tree plots in contrast to the other treatments (Supplemental Figure 2). Mean tree species richness was significantly higher in the LFS-Tree treatment (5.0 ± 0.5) than the Tree-Only (3.1 ± 0.4), and Untreated plots (1.8 ± 0.4; $F_{3,28} = 6.85$, $p < 0.0l$; TukeyHSD $p < 0.05$). LFS-Only tree species richness (3.1 ± 0.6) did not significantly differ from any other treatment. Jack pine and white birch were the two dominant species in the LFS-Tree and Tree-Only treatments (Supplemental Table 2; Supplemental Figure 3). Jack pine represented on average 52% of trees in LFS-Tree plots and 73% of trees in Tree-Only treatments and white birch represented on average 40% and 22% of trees in LFS-Tree plots and Tree-Only plots, respectively (Supplemental Table 2). White birch was the dominant tree species in the unplanted plots representing on average 62% and 78% in the Untreated and LFS-Only treatments, respectively. Importantly, species counts revealed that the number of planted jack pine was not significantly different between the LFS-Tree and Tree-Only treatments (Supplemental Figure 2).

Fine soil bulk density ranged from 0.5 to 1.1 g cm^{-3} and was generally consistent among the treatments with an overall mean of 0.79 ± 0.03 g cm^{-3}, except that bulk density in LFS-Tree (0.68 ± 0.04 g cm^{-3}) plots was significantly lower than LFS-Only plots (0.92 ± 0.06 g cm^{-3}; $F_{3,28} = 5.67$, $p < 0.01$; TukeyHSD $p < 0.01$; Supplemental Figure 4).

Soil pH was highly variable across plots and treatments and mean plot level pH values for mineral soil at a depth of 0–5 cm ranged from 4.2 to 6.5 ($F_{3,28} = 18.29$, $p < 0.001$; Supplemental Table 2). Liming significantly increased soil pH on average from 4.35 ± 0.03 (non-limed plots) to 4.93 ± 0.1 (limed plots; TukeyHSD $p < 0.001$; Supplemental Table 2). There was no significant difference in pH between the two limed treatments (LFS-Only and LFS-Tree), nor between the two non-limed treatments (Tree-Only and Untreated).

The mean mineral soil depth (sampled to bedrock) was 11.0 ± 0.8 cm, but pockets of deeper soil were located across the plots with the maximum depth of 50 cm (the depth of the soil probe) recorded.

3.2. Carbon pools

Total ecosystem carbon storage varied from 21.4 Mg ha^{-1} to 124.5 Mg ha^{-1} across all plots (mean 67.6 ± 4.8 Mg ha^{-1}; Supplemental Table 2) but no significant difference in total ecosystem carbon was observed among the treatments (Supplemental Figure 3).

3.2.1. Tree carbon

The tree carbon pool ranged from 1% of total ecosystem carbon in Untreated plots to 26% in LFS-Tree plots (Supplemental Table 2). Tree planting significantly increased the tree carbon pool by ca. 27 times in the Tree-Only plots compared to the Untreated plots and by ca. 7 times in the LFS-Tree plots compared with LFS-Only plots ($F_{3,28} = 40.56$, $p < 0.001$; TukeyHSD $p < 0.01$; Table 1). The combination of soil amendments and tree-planting resulted in a significant increase in the tree carbon pool, which was ca. 1.5 times greater in the LFS-Tree compared with Tree-Only plots (TukeyHSD $p < 0.01$). Interestingly, the soil amendment increased the amount of carbon stored per tree with individual jack pine trees in the LFS-Tree plots storing on average 1.5 times more carbon than jack pine in the Tree-Only plots ($t_{1486.2} = 11.66$, $p < 0.001$; Figure 3a). Similarly, white birch in the LFS-Only plots stored ca. 1.7 times more carbon compared with white birch in the Untreated plots ($F_{3,1529} = 4.51$, $p < 0.01$; TukeyHSD $p < 0.05$; Figure 3b).

3.2.2. Mineral soil organic carbon

Mineral soil (to a maximum depth of 20 cm) was the largest carbon pool across all treatments, with SOC comprising 79% of carbon in Untreated plots, 75% in LFS-Only plots, 64% in Tree-Only plots, and 57% in LFS-Tree plots (Table 1; Supplemental Table 2). However, there was no significant difference in SOC (Mg ha^{-1}) among the treatments. Similarly, carbon concentration did not significantly differ within each measured soil layer among the treatments; except that the carbon concentration in LFS-Tree plots was significantly higher than the Tree-Only plots in the 0–5 cm ($F_{3,28} = 4.51$, $p < 0.05$; TukeyHSD $p < 0.05$) and 5–10 cm layer ($F_{3,28} = 3.70$, $p < 0.05$; TukeyHSD $p < 0.05$; Figure 4). Soil carbon concentrations did not significantly differ with soil depth within any treatment.

3.2.3. Forest floor (LFH), understory, and downed woody debris carbon

Forest floor (LFH) was the third largest carbon pool among the treatments, representing 12% of total carbon on average (Table 1; Supplemental Table 2). LFH carbon storage was 50% higher in LFS-Tree plots than Tree-Only plots ($F_{3,28} = 4.95$, $p < 0.01$; TukeyHSD $p < 0.05$; Table 1). Understory (shrubs and herbs) carbon pools generally represented 5% of total carbon, with herbs storing 3% of total carbon on average, and shrubs storing 2% of total carbon on average (Table 1; Supplemental Table 2). Restoration treatment did not affect shrub carbon storage or herb carbon storage. Downed woody debris comprised relatively small carbon pools. Fine woody debris carbon (averaged 0.4% of total carbon) was unaffected by restoration treatments (Table 1; Supplemental Table 2). Coarse woody debris stored 0.1% of total carbon on average, and no differences were observed among the treatments (Table 1; Supplemental Table 2).

Table 1. Mean (± SE) carbon (Mg ha^{-1}) stored in each carbon pool for Untreated ($n = 5$), LFS-Only ($n = 7$), Tree-Only ($n = 8$), and LFS-Tree ($n = 12$) plots. Superscript letters indicate significance differences among treatments at $p < 0.001$ for trees and $p < 0.05$ for forest floor (LFH). No significant differences were observed among the other carbon pools.

Treatment	**Tree**	**Forest floor (LFH)**	**SOC**	**Shrub**	**Herb**	**Coarse woody debris**	**Fine woody debris**
Untreated	0.51^{a} (0.40)	6.37^{ab} (0.63)	45.45 (11.64)	0.75 (0.37)	2.65 (0.83)	0.07 (0.02)	0.17 (0.04)
LFS-Only	2.81^{a} (1.81)	6.43^{ab} (0.73)	43.00 (11.64)	0.95 (0.85)	2.48 (0.75)	0.01 (0.004)	0.17 (0.06)
Tree-Only	13.65^{b} (0.86)	5.96^{a} (0.57)	44.17 (4.95)	2.21 (1.36)	1.81 (0.32)	0.08 (0.03)	0.36 (0.05)
LFS-Tree	20.19^{c} (1.50)	8.94^{b} (0.66)	45.94 (5.43)	1.73 (0.53)	1.68 (0.41)	0.08 (0.02)	0.28 (0.05)

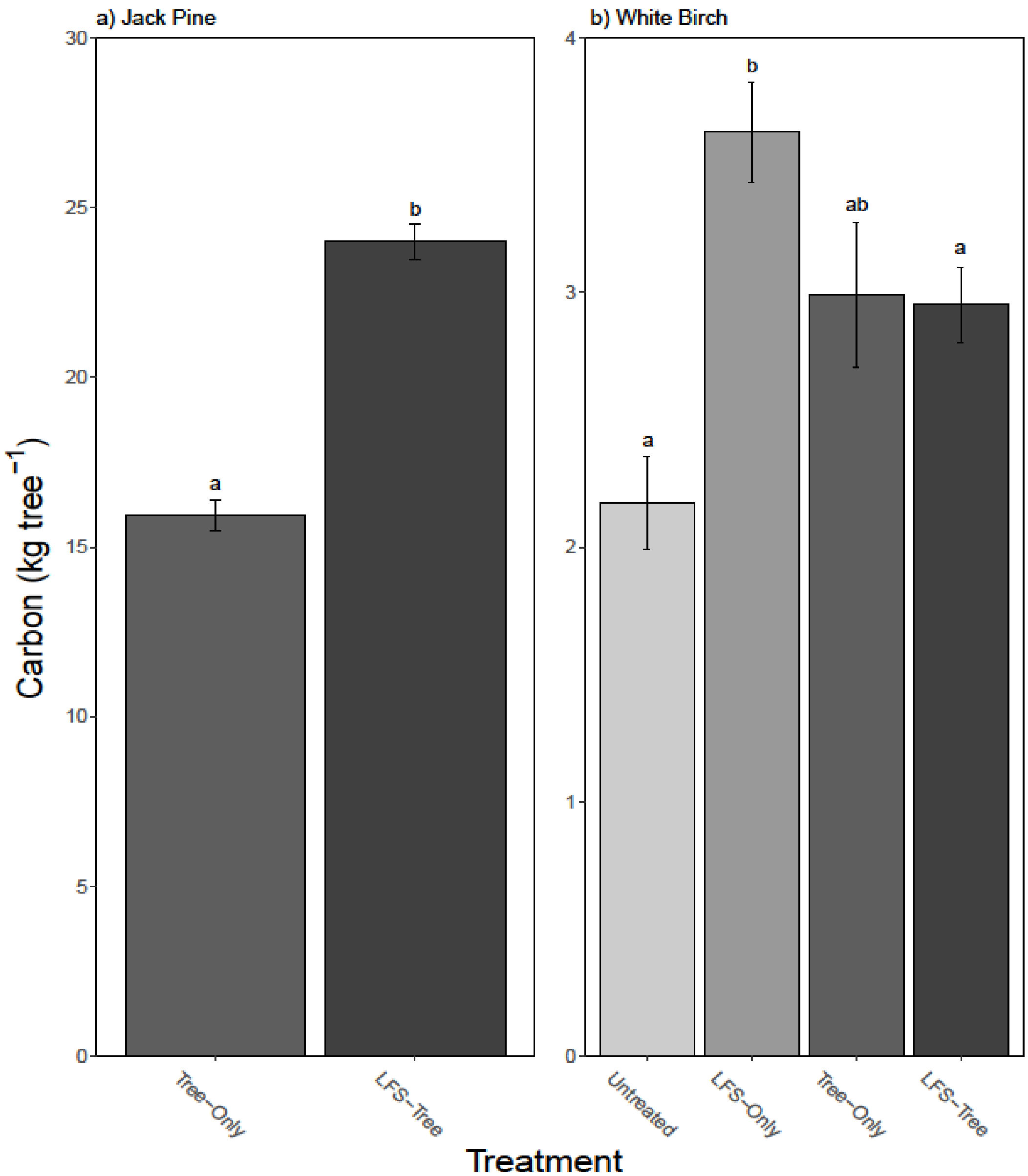
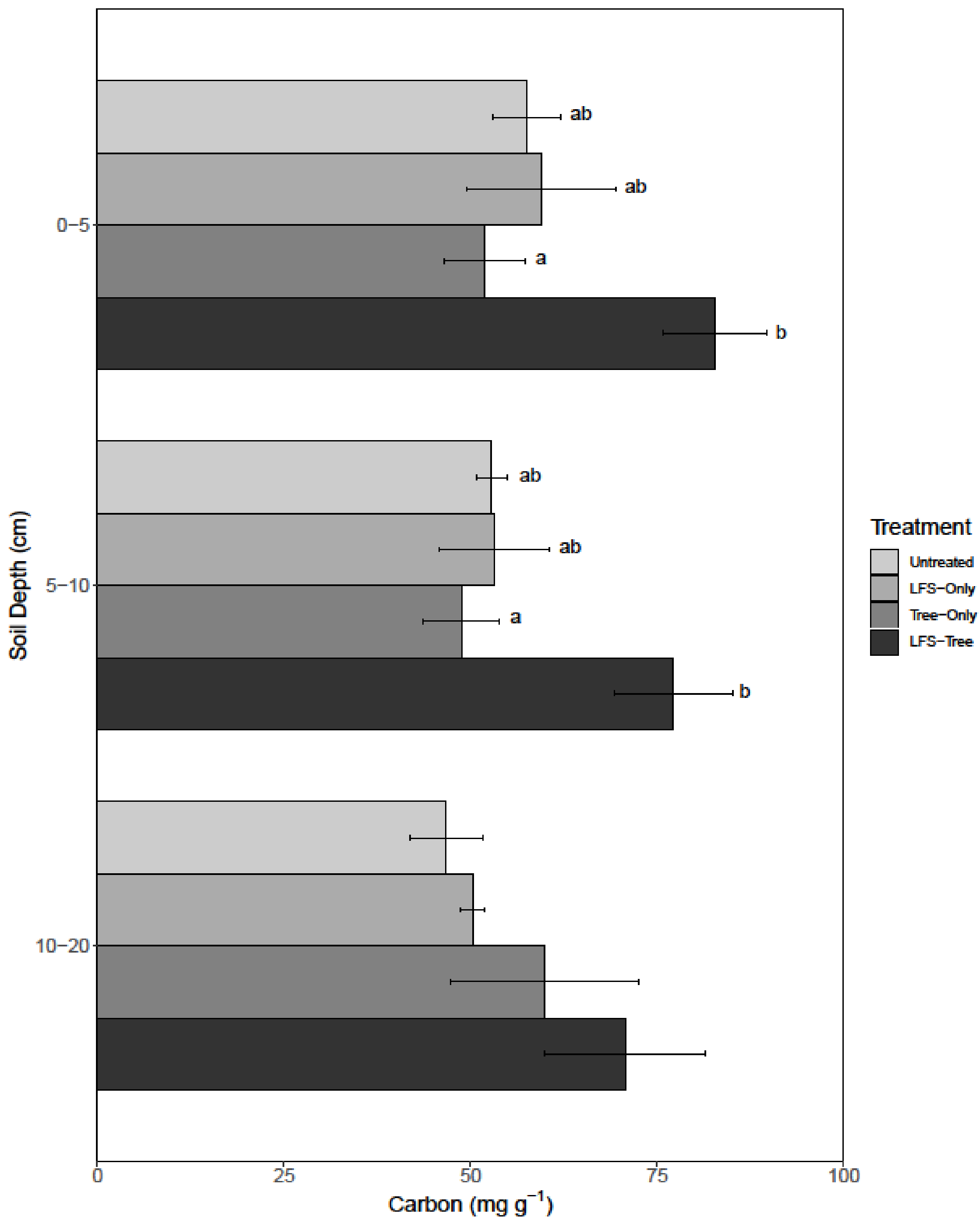


Figure 3. Mean (± SE) carbon (kg tree^{-1}) stored in (a) jack pine in Tree-Only (n = 8), and LFS-Tree (n = 12) plots and (b) white birch in Untreated (n = 5), LFS-Only (n = 7), Tree-Only (n = 8), and LFS-Tree (n = 12) plots. Jack pine was only compared between Tree-Only and LFS-Tree plots because it was not planted in LFS-Only or Untreated plots. Different letters indicate a difference at $p < 0.001$ between treatments for jack pine and $p < 0.05$ among treatments for white birch.

419 

420 **Figure 4.** Mean (± SE) soil organic carbon concentration (mg g^{-1}) with depth among different

421 restoration treatments: Untreated (n = 5), LFS-Only (n = 7), Tree-Only (n = 8), and LFS- Tree (n

422 = 12). Different letters indicate a difference at $p < 0.01$ in the 0–5 cm soil layer and at $p < 0.05$ in

423 the 5–10 cm soil layer.

3.3. Soil metal concentrations

The overall median among all treatments for total concentrations of chemicals of concern exceed the provincial guidelines for arsenic (7 times), cadmium (6.3 times), copper (10 times), nickel (4.7 times), and selenium (7.2 times), whereas cobalt (0.8 times) and lead (0.4 times) were lower than the provincial standard (Ontario Ministry of the Environment Conservation and Parks, 2011; Table 2). There was no significant difference in the total concentration of these chemicals of concern among the treatments, except for selenium where lower concentrations were observed in the Untreated plots compared with the other treatments ($\chi^2_3 = 12.90$, $p < 0.01$; Dunn's $p < 0.05$; Table 2). Only total selenium significantly correlated with tree carbon (Mg ha^{-1}; $r = 0.5$, $S = 2845.8$, $p < 0.01$).

Bioavailable concentrations were below detection limits for arsenic, cadmium, and selenium samples as well as for the majority (> 80%) of lead samples. Bioavailable cobalt was approximately 3 times lower in the LFS-Tree treatment compared with the Tree-Only treatment ($\chi^2_3 = 10.12$, $p < 0.05$; Dunn's $p < 0.05$; Table 3). Bioavailable copper was ca. 3.16 times greater in the Tree-Only treatment compared with LFS-Only treatment ($\chi^2_3 = 10.08$, $p < 0.05$; Dunn's $p < 0.05$). Bioavailable nickel was ca. 4 times higher in both un-limed (Untreated and Tree-Only) treatments compared with limed treatments (LFS-Only and LFS-Tree; $\chi^2_3 = 14.64$, $p < 0.01$; Dunn's $p < 0.05$; Table 3). No other treatment differences were observed. Bioavailable cobalt and nickel were significantly correlated with SOC (Mg ha^{-1}; cobalt: $S = 3202.7$, $p < 0.05$; nickel: $S = 3490.7$, $p < 0.05$), but no other significant correlations were observed.

Table 2. Total concentrations (median, IQR) of chemicals of concern (mg kg^{-1}) for Untreated (n = 5), LFS-Only (n = 7), Tree-Only (n = 8), and LFS-Tree (n = 12) plots. Concentrations below detection limits were set as half of detection limit to calculate median; detection limits (mg kg^{-1}) were 0.2 for arsenic, 0.05 for cadmium, 0.03 for cobalt, 0.05 for copper, 0.2 for nickel, 0.06 for lead, and 0.6 for selenium. Superscript letters indicate significance differences among treatments at $p < 0.05$ for selenium. Provincial soil standards were based on non-agricultural soils in Ontario, Canada (Ontario Ministry of the Environment Conservation and Parks, 2011).

Treatment	**Arsenic**	**Cadmium**	**Cobalt**	**Copper**	**Lead**	**Nickel**	**Selenium**
Untreated	127 (41)	6.94 (0.98)	16.5 (0.8)	975 (134)	49.5 (5.2)	395 (87)	5.35 (0.98)^{a}
LFS-Only	121 (34.5)	7.65 (1.66)	16.8 (4.75)	856 (318.5)	40.8 (12.9)	368 (68)	9.91 (2.77)^{b}
Tree-Only	110 (35.93)	7.91 (2.32)	16.4 (1.93)	853 (244.5)	39.6 (15.27)	359.5 (84.25)	12.4 (2.44)^{b}
LFS-Tree	143.5 (89.75)	7.41 (2.3)	21.5 (8.13)	1085 (616)	51.75 (30.28)	500 (299.75)	11.75 (2.36)^{b}
Provincial soil standard	18	1.2	21	92	120	82	1.5

Table 3. Bioavailable concentrations (median, IQR) of chemicals of concern (μg kg^{-1}) for Untreated (n = 5), LFS-Only (n = 7), Tree-Only (n = 8), and LFS-Tree (n = 12) plots. Arsenic, cadmium, and lead were below detection limits. Concentrations below detection limits were set as half of detection limit to calculate median; detection limits (mg kg^{-1}) were 0.03 for cobalt, 0.05 for copper, and 0.2 for nickel. Superscript letters indicate significance differences for each element among the treatments at $p < 0.05$.

Treatment	Cobalt	Copper	Nickel
Untreated	$0.06 (0.01)^{ab}$	$1.05 (0.3)^{ab}$	$1.29 (0.87)^{a}$
LFS-Only	$0.02 (0.03)^{ab}$	$0.7 (0.76)^{a}$	$0.27 (0.42)^{b}$
Tree-Only	$0.06 (0.03)^{a}$	$2.22 (1.64)^{b}$	$1.19 (0.37)^{a}$
LFS-Tree	$0.02 (0)^{b}$	$1.09 (1.07)^{ab}$	$0.34 (0.37)^{b}$

4. Discussion

4.1. Influence of restoration treatment on tree carbon storage

We demonstrate that soil amendments increased tree carbon sequestration on an acid and metal impacted landscape in Sudbury, Ontario, Canada. But in these severely damaged upland sites the tree carbon pool of the tree-planted and soil amended treatment (LFS-Tree) was much smaller (ca. < 4 Mg ha^{-1}) compared with similarly aged Sudbury stands reported in Preston et al. (2020), who reported a mean tree carbon pool of 23.9 Mg ha^{-1} in 25-year-old stands. This difference occurred despite our LFS-Tree plots having more than double the tree density (median 1400 stems ha^{-1}) compared to the Preston et al. (2020) study (median 489 stems ha^{-1} in 25 year old stands). The greater stem density within our study could have resulted in lower individual biomass due to increased competition between trees (Looney et al., 2016). However, it is unlikely that stem density was high enough to inhibit carbon storage at our study plots. Hunt et al. (2010) reported aboveground tree carbon pools in jack pine stands planted at high densities (mean 2288 stems ha^{-1}) to be 3.3 times greater than in our plots. Furthermore, Park (2015) also found relatively higher aboveground tree carbon in similar-aged jack pine stands planted at comparatively higher densities (mean 2264 stems ha^{-1}). The more likely explanation for the lower tree carbon pool in our study, than in Preston et al. (2020), is that the upland study area (Daisy Lake watershed) is more exposed to the weather, resulting in slower tree growth (Eränen & Kozlov, 2006). The relatively smaller size of similar-aged planted trees in our restored plots (ca. 1.6 times smaller mean DBH) compared to Preston et al. (2020) corroborates this hypothesis.

Tree planting without soil amendment (Tree-Only) is currently the most common restoration technique across the Sudbury landscape (ca. 16,500 ha or 83% of the restored area; pers. comm. P. Beckett), due to its lower cost (ca. \$3,500 ha^{-1}) compared to a higher intensity treatments (LFS-Only, ca. \$17,000 ha^{-1}, LFS-Tree, ca. \$20,000 ha^{-1}; pers. comm. T. McCaffrey). Tree-Only restoration did result in a significant store of tree carbon (i.e., 13.1 Mg ha^{-1} above Untreated plots), but had a lower tree carbon pool (ca. 48%) compared with the soil amended plots. Jack pine density (and total tree density) was similar between the LFS-Tree and Tree-Only treatments in this study, thus the reduction in Tree-Only tree carbon storage demonstrates that the soil amendment (LFS) had a significant impact on tree growth and ultimately tree carbon sequestration, as indicated by the increased individual tree biomass with soil amendment. Tree-Only restoration is a relatively cost-effective restoration approach for tree carbon sequestration, but conservation managers must recognize the reduction in tree carbon sequestration compared with the LFS-Tree treatment.

Tree carbon sequestration via natural tree regeneration in the Untreated and soil amended plots without tree planting (LFS-Only) was similar but only represented ca. 8% of the tree pool in the full treatment of soil amendments with tree planting (LFS-Tree). This was primarily because both unplanted treatments were dominated by white birch that has a lower carbon content (ca. 7 times) compared with jack pine and due to the significantly lower stand density (ca. 5.5 times) compared to the planted plots. Plot level white birch carbon content increased in the LFS-Only treatment indicating that soil amendment stimulates tree growth, consistent with previous studies (Winterhalder, 1996). This increase in white birch carbon was not observed between Tree-Only and LFS-Tree treatments probably because of competition from jack pine limiting white birch growth. Previous studies in the region have indicated that recolonization by tree species is primarily due to seed dispersal and not regeneration of the seed bank (Winterhalder, 1995); hence the dominance of white birch, which is wind dispersed (Hughes & Fahey, 1988). Stand density was highly variable within LFS-Only and Untreated plots indicating that overall dispersal is slow and not consistent across the landscape even when soil conditions are improved. Ultimately, recolonization is dependent on the proximity to other established stands that can act as a seed source (Bartels et al., 2016). Thus, natural regeneration on this landscape is not a dependable restoration strategy with respect to carbon sequestration because the dispersal kernel of trees surrounding Sudbury’s barren area was exceeded by the scale of disturbance. Some of the planted jack pine trees are now seed producing, so the planted areas are likely now able to disperse seeds into non-planted areas and further benefit tree carbon storage in subsequent jack pine generations. Jack pine cones are serotinous and require high temperatures (often from forest fires) to open, however seeds in sun-exposed portions of the tree can open in the absence of fire (Burns & Honkala, 1990). Surface temperatures in our jack pine planted plots exceeded 50°C on hot summer days (unpublished data), which is the melting point of the serotinous cones and would enable seed dispersal (Burns & Honkala, 1990). Ultimately, jack pine has stabilized these degraded sites, but to maintain the long-term viability of the ecosystem conservation managers should increase the tree species diversity.

Taken together these findings highlight that restoration treatment, initial soil conditions, site exposure, and tree dispersal all play significant roles in determining tree biomass accumulation and therefore carbon sequestration potential. Thus, studies that attempt to estimate carbon storage following global tree restoration (Griscom et al., 2017; Fuss et al., 2018; Bastin et al., 2019) should consider these factors in order to provide realistic carbon storage estimation (Skidmore et al., 2019). The most intensive restoration treatments (LFS-Tree) also had greater tree species richness than plots that were only tree-planted (Tree-Only), which should result in more ecosystem services such as amelioration of weather (e.g.,, providing wind breaks), soil detoxification, soil formation, and provision of food and wood (Díaz & Cabido, 2001; Lewis et al., 2019). Future studies should investigate how well trees survive in the Tree-Only treatment over a longer time period, by monitoring age at senescence and death compared to the full LFS-Tree treatments. Jack pine seedlings are generally more affected by pollution than mature trees (Armentano & Menges, 1987). However Beckett et al. (1995; as cited in Winterhalder, 1996) found that metal content in Sudbury jack pine trees increases with age, so the metal effects could cause premature senescence in the future.

4.2. Restoration treatment effect on other carbon pools

Restoration treatment had no significant impact on the total ecosystem carbon pool, primarily because the SOC pool was the largest carbon pool and was also highly variable across the plots (14–91 Mg ha^{-1}); consistent with previous studies in the area (Hazlett et al., 1984; Preston et al., 2020). This variability in SOC is probably due to the largescale erosion processes that have taken place on this upland landscape over the past century that buried organic-rich soils or exported it to lakes and other receiving waters (Hazlett et al., 1984). Indeed, we observed evidence of this phenomenon as we identified small fragments of charcoal and low bulk densities at depths greater than 10 cm, indicating the presence of organic matter. Despite the variability, the average SOC pool across all treatments was similar to that reported in other jack pine dominated stands in Ontario (Foster et al., 1995; Hunt et al., 2010; Preston et al., 2020). Repeated measurement of the SOC pool is required to determine changes over time and with treatment (Smith, 2004; Schrumpf et al. 2011; Preston et al., 2020). Therefore, we assessed changes between the treatments in terms of SOC concentration. Interestingly, SOC concentration was only significantly greater than the untreated soils in the 0–5 cm and 5–10 cm layers in the LFS-Tree treatment. This treatment also had the greatest average tree species richness and all thirteen identified trees were found in this LFS-Tree treatment, consistent with other studies in the region (Nkongolo et al., 2013, 2016). A positive relationship between SOC pool and species diversity has been identified within temperate and boreal forests in Sweden (Gamfeldt et al., 2013), which could be due to several mechanisms such as above and belowground complementarity, higher aboveground litter inputs to soils, and higher belowground litter inputs (reviewed in Mayer et al. 2020). However, this increase in SOC concentration was not observed further down the soil profile suggesting carbon may be being mineralized by increased microbial activity with the improved soil conditions (Narendrula-Kotha & Nkongolo, 2017).

The effect of treatment on SOC concentration and pools warrants further investigation as this is a sizeable carbon pool on the landscape and its stability will influence restoration efforts that attempt to maximize future carbon sequestration.

The LFH pool was also a noteworthy store of carbon (ca. 12%), consistent with other studies (Hunt et al., 2010; Park, 2015; Preston et al., 2020), but surprisingly there was no difference between untreated and restored plots, likely because acidic, metal polluted soils in un-amended plots decreased litter decomposition rates (Johnson & Hale, 2004). In addition, overland flow of litter during storm events may redistribute the LFH layer. Downed woody debris (both coarse and fine) was a very small carbon pool (ca. < 1% of total ecosystem carbon) in contrast to mature forests where it comprises approximately 20% of total carbon storage (Russell et al., 2015). This is probably due to the relatively young age of the plots (24 years) with downed woody debris expected to increase in size as the stands mature (Keith et al., 2009). Restoration did not affect herb and shrub carbon in our study. This was probably due to the increased tree growth with restoration that subjected understory plants to light competition, preventing a similar increase in understory carbon (Burton et al., 2013).

4.3. Influence of restoration on soil pH and metal concentrations

Soil liming has been previously demonstrated to improve both the survival and growth of tree seedlings in smelter-impacted industrial barrens in Russia (Eränen and Kozlov, 2006), although Kozlov and Haukioja (1999) found successful survival of planted seedlings in smelter-impacted industrial barrens without the use of liming or fertilizing. Increasing soil pH should reduce the bioavailability of trace metals allowing for seedling establishment (Bolan et al., 2003). In Sudbury, Nkongolo et al. (2013) showed that historically limed areas continue to have a greater pH (ca. pH 6) and have lower bioavailable metal concentrations than un-limed areas (ca. pH 4) after 20–30 years of treatment. In contrast, soil pH within the limed plots in the present study was on average only ca. 0.5 units greater than the untreated plots suggesting the neutralizing power of the dolomitic limestone is close to exhaustion, and/or that continued soil erosion has reduced the buffering effect of the aerial treatment.

Total concentrations of arsenic, cadmium, copper, nickel and selenium (i.e., chemicals of concern) were higher than provincial standards (Ontario Ministry of the Environment Conservation and Parks, 2011) and consistent with other studies conducted with the Sudbury region (Meadows & Watmough, 2012; Nkongolo et al., 2013). Total observed metal concentration did not vary with restoration treatment probably due to the similar soil pH and SOC concentration among the treatments; two parameters known to influence metal concentration in soils (Meadows & Watmough, 2012). However, total selenium concentration was significantly lower in the Untreated plots and thus positively correlated with tree carbon. One possible explanation for this observation is that the presence of the tree canopy (which we assumed to increase with greater tree carbon pool) may have led to greater selenium deposits to the soil via increased throughfall deposition compared with rainfall alone (Roulier et al., in press). Alternatively, varied soil movement and erosion across plots (Supplemental Figure 1) may have unevenly distributed metals resulting in no clear trends. However, neither of these mechanisms fully explain other observed metals concentrations at our study plots.

Bioavailable metals were measured to estimate the potential effect of restoration treatment on soil toxicity. Only three metals were observed above the instrument detection limit (cobalt, copper, and nickel) at low concentrations similar to those previously reported within Sudbury region; thus, indicating that potential phytotoxicity among the treatments is low (Nkongolo et al., 2013). Overall, there was no consistent trend among the three bioavailable metals among the treatments. However, bioavailable cobalt and nickel were correlated with SOC (Mg ha^{-1}) across the watershed, likely because both metals form complexes with organic matter that render them more bioavailable when organic carbon is increased (Lange et al., 2016; Sunet al., 2018). Nickel bioavailability was also lower in treatments that had been limed (LFS-Only and LFS-Tree) presumably due to greater base cation concentration in the soils (Sreekanth et al., 2013). As outlined above, the effect of the dolomitic limestone may be waning thus nickel concentrations in the limed treatments could increase in the near future. Although, this is not expected to cause significant stress to trees as indicated by the continued survival of trees in the Tree-Only treatment.

5. Conclusion

Amending degraded soil with lime and fertilizer resulted in larger trees and significantly increased tree carbon sequestration rates within planted forest plots on an acid and metal impacted landscape. Tree planting alone also increased tree carbon storage compared to untreated plots. Conservation and land managers should thus weigh carbon sequestration opportunities with the financial costs of soil amendments to meet their restoration goals. At these exposed upland sites, even combined tree restoration and soil amendments resulted in a smaller tree carbon pool compared to other sites on the same landscape, highlighting that other environmental variables such as pollution gradients, the extent of past erosion, and tree dispersal all play significant roles in determining tree biomass accumulation and carbon sequestration potential. This study demonstrates that it is essential to consider restoration treatment and land history when estimating carbon sequestration potential both on an individual landscape and at a global scale. Greater emphasis on soil building processes is also needed because this dominant carbon pool appears extremely slow to respond to remediation efforts.

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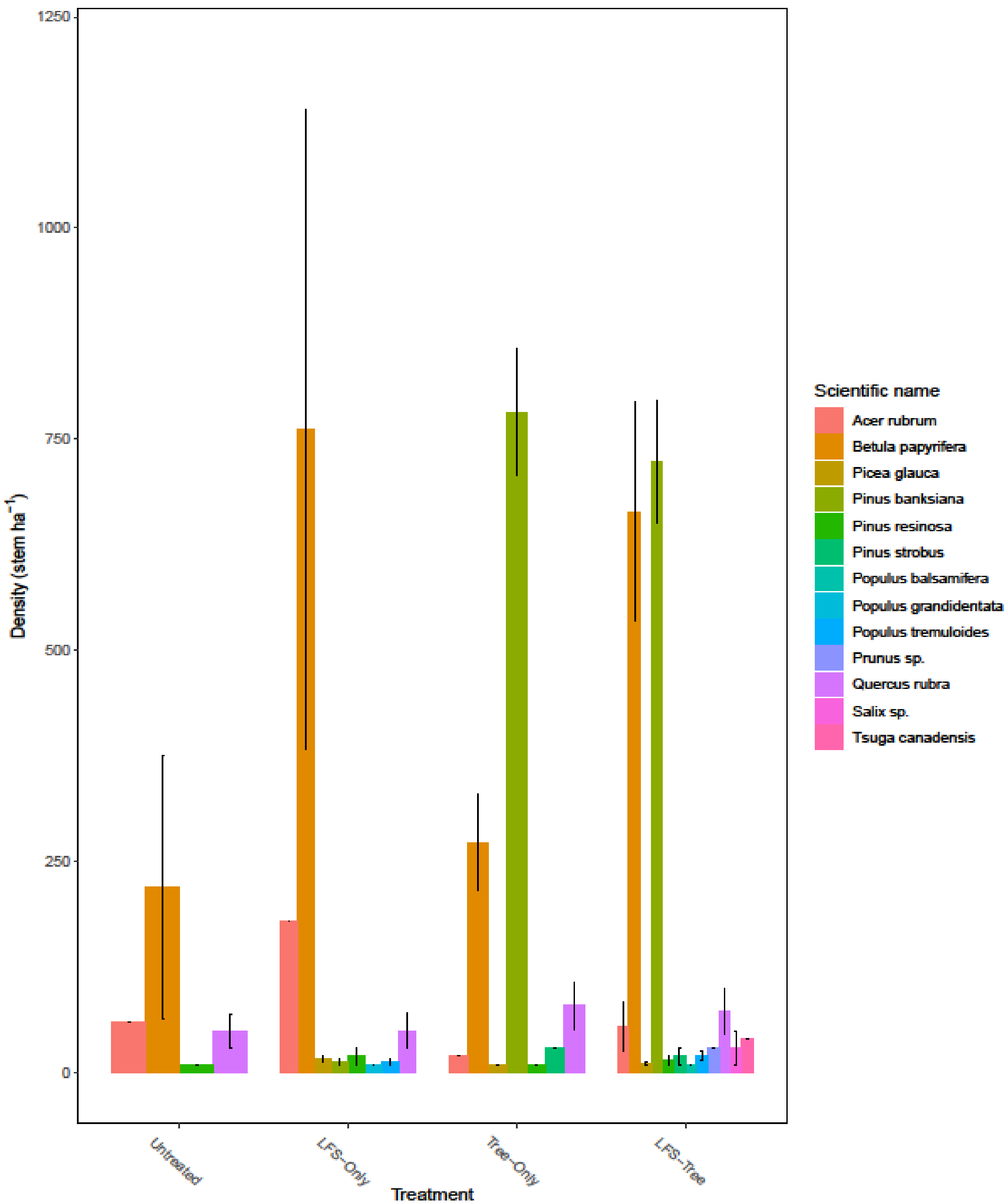
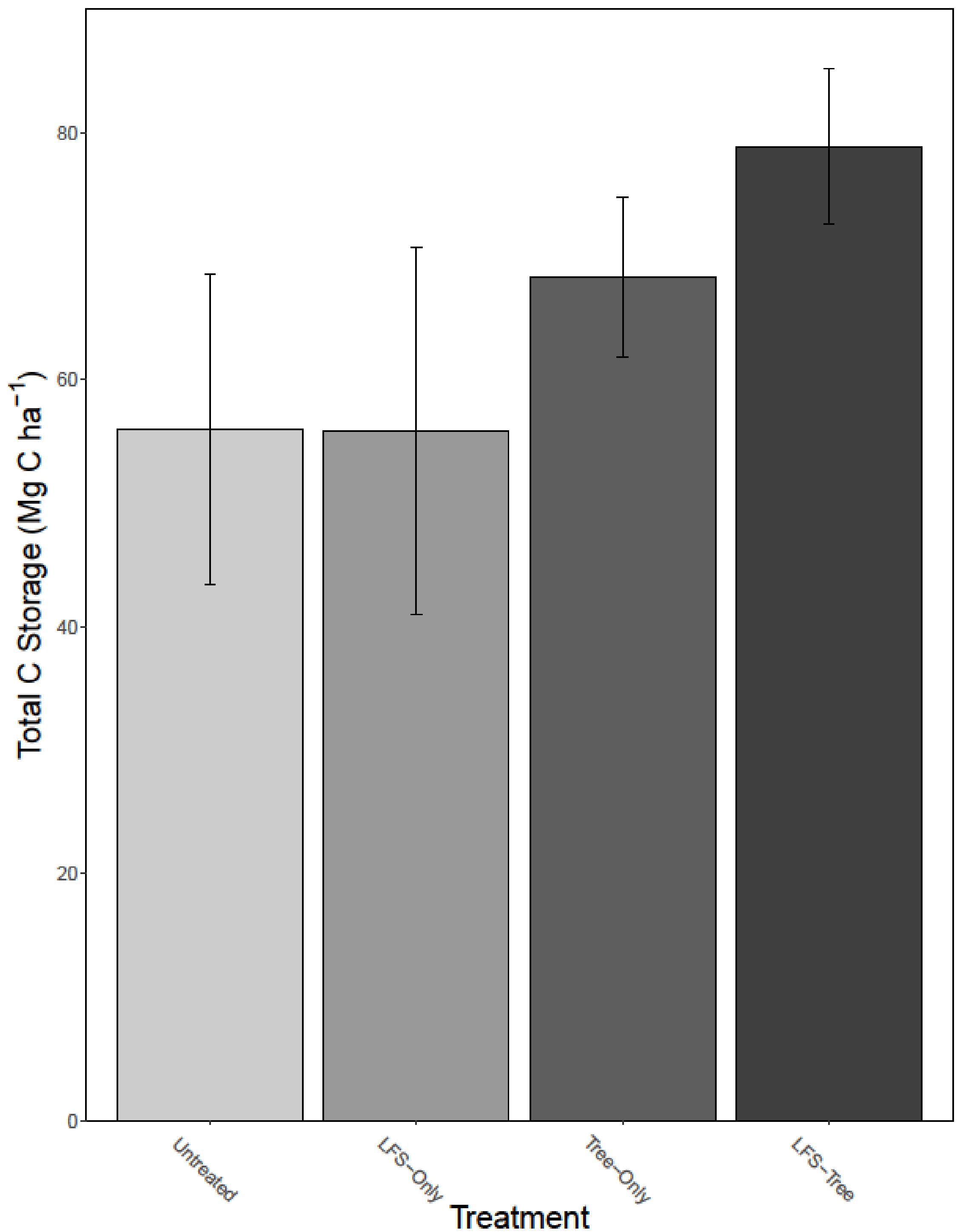
7. Supplemental data

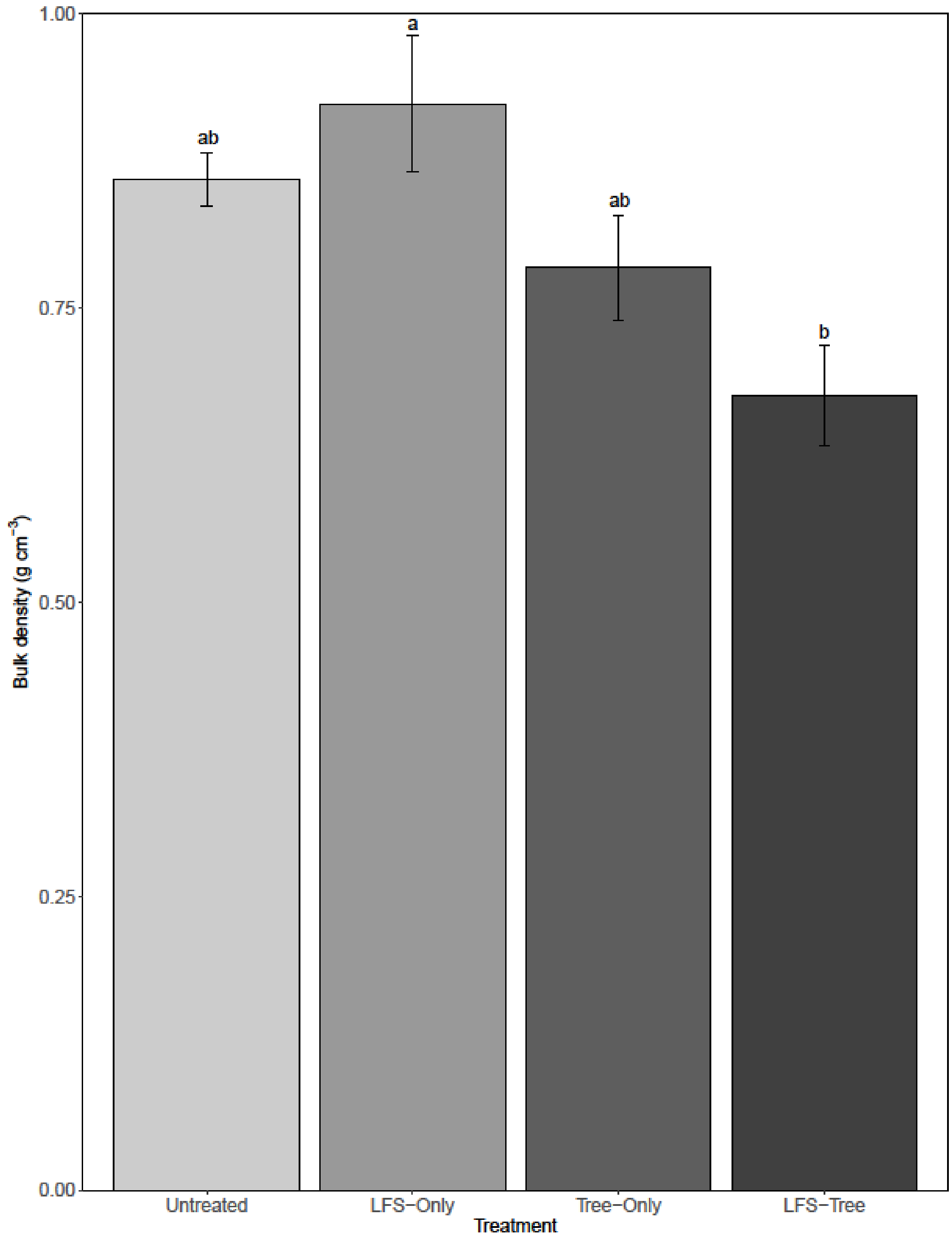
Supplemental Figure 1. Photos of the upland areas of the Daisy Lake watershed showing (A) and (B) the eroded bedrock outcrops with pockets of soil between bedrock and (C) a petrified stump with exposed roots indicating soil loss from this location.



Supplemental Figure 2. Density of identified tree species (mean ± SE) among different restoration treatments (n = 32): Untreated (n = 5), LFS-Only (n = 7), Tree-Only (n = 8), and LFS-Tree (n = 12) plots.



Supplemental Figure 3. Total ecosystem carbon storage (mean ± SE) among Untreated (n = 5), LFS-Only (n = 7), Tree-Only (n = 8), and LFS-Tree (n = 12) plots.



Supplemental Figure 4. Mineral soil (all depths pooled) bulk densities (mean ± SE) among restoration treatments: Untreated (n = 5), LFS-Only (n = 7), Tree-Only (n = 8), and LFS-Tree (n = 12) plots. Different letters indicate a difference at $p < 0.01$ among the restoration treatments.

Supplemental Table 1. Summary statistics for analyses performed. All tests were performed at a significance level of 0.05.

Statistical Test	**Summary Statistics**
Kruskal-Wallis: tree stem density ~ treatment	Kruskal-Wallis chi-squared = 17.28, df = 3, p-value = 0.0006
Dunn's test: tree density ~ treatment	LFS-Only vs. LFS-Tree p adj = 0.01 LFS-Only vs. Tree-Only p adj 0.21 LFS-Tree vs. Tree-Only p adj 0.22 LFS-Only vs. Untreated p adj 0.27 LFS-Tree vs. Untreated p adj 0.001 Tree-Only vs. Untreated p adj 0.04
ANOVA: tree species richness ~ treatment	$F_{3,28}$ = 6.85 p = 0.0013
TukeyHSD: tree species richness ~ treatment	LFS-Tree vs. LFS-Only p adj = 0.053 Tree-Only vs. LFS-Only p adj = 0.10 Untreated vs. LFS-Only p adj = 0.40 Tree-Only vs. LFS-Tree p adj = 0.04 Untreated vs. LFS-Tree p adj = 0.001 Untreated vs. Tree-Only p adj = 0.39
T-test: number of jack pine ~ treatment (LFS-Tree vs. Tree-Only)	t = -0.56, df = 16.67, p = 0.58
ANOVA: soil bulk density ~ treatment	$F_{3,28}$ = 5.67, p = 0.004
TukeyHSD: soil bulk density ~ treatment	LFS-Only vs. Untreated p adj = 0.84 Tree-Only vs. Untreated p adj = 0.76 LFS-Tree vs. Untreated p adj = 0.07 Tree-Only vs. LFS-Only p adj = 0.21 LFS-Tree vs. LFS-Only p adj = 0.003 LFS-Tree vs. Tree-Only p adj = 0.30
ANOVA: soil pH ~ treatment	$F_{3,28}$ = 18.29, p = 9.07e-07
TukeyHSD: soil pH ~ treatment (LFS-Only vs. LFS-Tree)	LFS-Only vs. Untreated p adj = 0.0004 Tree-Only vs. Untreated p adj = 0.81 LFS-Tree vs. Untreated p adj = 0.00002 Tree-Only vs. LFS-Only p adj = 0.001 LFS-Tree vs. LFS-Only p adj = 0.90 LFS-Tree vs. Tree-Only p adj = 0.00003
ANOVA: total carbon ~ treatment	$F_{3,28}$ = 1.48, p = 0.24
ANOVA: tree carbon ~ treatment	$F_{3,28}$ = 40.56, p = 2.53e-10
TukeyHSD: tree carbon ~ treatment (Untreated vs. Tree-Only)	LFS-Only vs. Untreated p adj = 0.78 Tree-Only vs. Untreated p adj = 0.00003 LFS-Tree vs. Untreated p adj = 0.0000000 Tree-Only vs. LFS-Only p adj = 0.0001 LFS-Tree vs. LFS-Only p adj = 0.0000000 LFS-Tree vs. Tree-Only p adj = 0.009
T-test: carbon per jack pine tree ~ treatment (Tree-Only vs. LFS-Tree)	t = 11.66, df = 1486.2, p-value < 2.2e-16
ANOVA: carbon per white birch tree ~ treatment	$F_{3,1529}$ = 4.51, p = 0.004
TukeyHSD: carbon per white birch tree ~ treatment	LFS-Tree vs. LFS-p adj = 0.02 Tree-Only vs. LFS-Only p adj = 0.26 Untreated vs. LFS-Only p adj = 0.01 Tree-Only vs. LFS-Tree p adj = 1.00 Untreated vs. LFS-Tree p adj = 0.32 Untreated vs. Tree-Only p adj = 0.40
ANOVA: SOC ~ treatment	$F_{3,28}$ = 0.03, p = 0.99
ANOVA: log(carbon concentration per 0-5 cm soil layer) ~ treatment	$F_{3,28}$ = 4.51, p = 0.01

TukeyHSD: log(carbon concentration for 0-5 cm soil layer) ~ treatment	LFS-Only vs. Untreated p adj = 1.00 Tree-Only vs. Untreated p adj = 0.89 LFS-Tree vs. Untreated p adj = 0.18 Tree-Only vs. LFS-Only p adj = 0.94 LFS-Tree vs. LFS-Only p adj = 0.07 LFS-Tree vs. Tree-Only p adj = 0.01
ANOVA: log(carbon concentration for 5-10 cm soil layer) ~ treatment	$F_{3,28} = 3.70$, $p = 0.02$
TukeyHSD: carbon concentration for 5-10 cm soil layer ~ treatment	LFS-Only vs. Untreated p adj = 0.99 Tree-Only vs. Untreated p adj = 0.92 LFS-Tree vs. Untreated p adj = 0.26 Tree-Only vs. LFS-Only p adj = 0.98 LFS-Tree vs. LFS-Only p adj = 0.09 LFS-Tree vs. Tree-Only p adj = 0.03
ANOVA: log(carbon concentration for 10-20 cm soil layer) ~ treatment	$F_{3,28} = 1.02$, $p = 0.40$
ANOVA: (Untreated) soil carbon concentrations ~ soil layer	$F_{2,12} = 1.80$, $p = 0.21$
ANOVA: (LFS-Only) soil carbon concentrations ~ soil layer	$F_{2,15} = 0.29$, $p = 0.75$
ANOVA: (Tree-Only) soil carbon concentrations ~ soil layer	$F_{2,20} = 0.50$, $p = 0.62$
ANOVA: (LFS-Tree) soil carbon concentrations ~ soil layer	$F_{2,30} = 0.48$, $p = 0.62$
ANOVA: Forest floor (LFH) carbon ~ treatment	$F_{3,28} = 4.95$, $p = 0.007$
TukeyHSD: Forest floor (LFH) ~ treatment	LFS-Only vs. Untreated p adj = 1.00 Tree-Only vs. Untreated p adj = 0.98 LFS-Tree vs. Untreated p adj = 0.08 Tree-Only-= vs. LFS-Only p adj = 0.97 LFS-Tree vs. LFS-Only p adj = 0.05 LFS-Tree vs. Tree-Only p adj = 0.01
Kruskal-Wallis: shrub carbon ~ treatment	Kruskal-Wallis chi-squared = 3.43, df = 3, p-value = 0.33
ANOVA: herb carbon ~ treatment	$F_{3,28} = 0.75$, $p = 0.53$
ANOVA: fine woody debris carbon ~ treatment	$F_{3,28} = 2.61$, $p = 0.07$
Kruskal-Wallis: coarse woody debris carbon ~ treatment	Kruskal-Wallis chi-squared = 9.47, df = 3, p-value = 0.02
Dunn's test: coarse woody debris carbon ~ treatment	LFS-Only vs. LFS-Tree -p adj = 0.07 LFS-Only vs. Tree-Only p adj = 0.06 LFS-Tree vs. Tree-Only p adj = 1.00 LFS-Only vs. Untreated -p adj = 0.08 LFS-Tree vs. Untreated p adj = 1.00 Tree-Only vs. Untreated p adj = 1.00
ANOVA: total arsenic ~ treatment	$F_{3,28} = 1.48$, $p = 0.24$
ANOVA: total cadmium ~ treatment	$F_{3,28} = 1.66$, $p = 0.20$
Kruskal-Wallis: total cobalt ~ treatment	Kruskal-Wallis chi-squared = 2.89, df = 3, p-value = 0.41
Kruskal-Wallis: total copper ~ treatment	Kruskal-Wallis chi-squared = 5.6, df = 3, p-value = 0.13
Kruskal-Wallis: total nickel ~ treatment	Kruskal-Wallis chi-squared = 3.54, df = 3, p-value = 0.32
Kruskal-Wallis: total lead ~ treatment	Kruskal-Wallis chi-squared = 4.74, df = 3, p-value = 0.19
Kruskal-Wallis: total selenium ~ treatment	Kruskal-Wallis chi-squared = 12.90, df = 3, p-value = 0.005
Dunn's test: total selenium ~ treatment	LFS-Only vs. LFS-Tree p adj = 0.57 LFS-Only vs. Tree-Only p adj = 0.49 LFS-Tree vs. Tree-Only p adj = 0.72 LFS-Only vs. Untreated p adj = 0.04

	LFS-Tree vs. Untreated p adj = 0.004 Tree-Only vs. Untreated p adj = 0.006
Kruskal-Wallis: bioavailable cobalt ~ treatment	Kruskal-Wallis chi-squared = 10.12, df = 3, p-value = 0.02
Dunn’s test: bioavailable cobalt ~ treatment	LFS-Only vs. LFS-Tree p adj = 0.57 LFS-Only vs. –Tree-Only p adj = 0.16 LFS-Tree vs. Tree-Only p adj = 0.04 LFS-Only vs. Untreated p adj = 0.19 LFS-Tree vs. Untreated p adj = 0.06 Tree-Only vs. Untreated p adj = 0.99
Kruskal-Wallis: bioavailable copper ~ treatment	Kruskal-Wallis chi-squared = 10.08, df = 3, p-value = 0.02
Dunn’s test: bioavailable copper ~ treatment	LFS-Only vs. LFS-Tree p adj = 0.51 LFS-Only vs. Tree-Only p adj = 0.02 LFS-Tree vs. Tree-Only p adj = 0.05 LFS-Only vs. Untreated p adj = 0.66 LFS-Tree vs. Untreated p adj = 0.85 Tree-Only vs. Untreated p adj = 0.08
Kruskal-Wallis: bioavailable nickel ~ treatment	Kruskal-Wallis chi-squared = 14.64, df = 3, p-value = 0.002
Dunn’s test: bioavailable nickel ~ treatment	LFS-Only vs. LFS-Tree p adj = 0.79 LFS-Only vs. Tree-Only p adj = 0.02 LFS-Tree vs. Tree-Only p adj = 0.010 LFS-Only vs. Untreated p adj 0.033 LFS-Tree vs. Untreated p adj = 0.037 Tree-Only vs. Untreated p adj = 0.79
Spearman’s rank correlation analysis: bioavailable copper ~ SOC	S = 3571.7, p-value = 0.05, r = 0.35
Spearman’s rank correlation analysis: bioavailable nickel ~ SOC	S = 3490.7, p-value = 0.04, r = 0.36
Spearman’s rank correlation analysis: bioavailable cobalt ~ SOC	S = 3202.7, p-value = 0.02, r = 0.41
Spearman’s rank correlation analysis: total cobalt ~ SOC	S = 6208.3, p-value = 0.45, r = -0.14
Spearman’s rank correlation analysis: total selenium ~ SOC	S = 4504.9, p-value = 0.34, r = 0.17
Spearman’s rank correlation analysis: total arsenic ~ SOC	S = 6030.1, p-value = 0.57, r = -0.11
Spearman’s rank correlation analysis: total copper ~ SOC	S = 4105.5, p-value = 0.17, r = 0.25
Spearman’s rank correlation analysis: total lead ~ SOC	S = 5042, p-value = 0.68, r = 0.08
Spearman’s rank correlation analysis: total nickel ~ SOC	S = 6668.2, p-value = 0.22, r = -0.22
Spearman’s rank correlation analysis: total cadmium ~ SOC	S = 5755, p-value = 0.77, r = -0.05
Spearman’s rank correlation analysis: total copper ~ tree carbon	S = 3922.9, p-value = 0.12, r = 0.28
Spearman’s rank correlation analysis: total arsenic ~ tree carbon	S = 4590, p-value = 0.39, r = 0.16
Spearman’s rank correlation analysis: total cobalt ~ tree carbon	S = 4218, p-value = 0.21, r = 0.23
Spearman’s rank correlation analysis: total cadmium ~ tree carbon	S = 5048.3, p-value = 0.68, r = 0.07
Spearman’s rank correlation analysis: total nickel ~ tree carbon	S = 4179.8, p-value = 0.20, r = 0.23
Spearman’s rank correlation analysis: total lead ~ tree carbon	S = 3899.4, p-value = 0.11, r = 0.29
Spearman’s rank correlation analysis: total selenium ~ tree carbon	S = 2845.8, p-value = 0.006, r = 0.48
Spearman’s rank correlation analysis: bioavailable cobalt ~ tree carbon	S = 6235.9, p-value = 0.44, r = -0.14
Spearman’s rank correlation analysis: bioavailable copper ~ tree carbon	S = 3929.2, p-value = 0.12, r = 0.28
Spearman’s rank correlation analysis: bioavailable nickel ~ tree carbon	S = 5672.8, p-value = 0.83, r = -0.04

Supplemental Table 2. Site characteristics and carbon (C; Mg ha^{-1}) stocks for each plot. Jack pine and white birch were the dominant tree species and therefore only their frequencies are specified. Soil pH is for the 0–5 cm mineral soil layer.

								Carbon pools							
Site Code	**Treatment**	**Tree Species Richness**	**Jack Pine Frequency**	**White Birch Frequency**	**Total Tree Frequency**	**Mean (± SE) Soil Depth (cm)**	**Soil pH**	**Tree**	**Shrub**	**Herb**	**CWD**	**FWD**	**SOC**	**LFH**	**∑ Total**
IP1MF	Untreated	2	0	2	3	10.3 (2.9)	4.3	0.1	1.0	4.5	0.0	0.1	40.4	6.3	52.5
IP2MF	Untreated	1	0	13	13	8.6 (2.2)	4.5	0.2	0.7	1.5	0.0	0.3	31.0	5.0	38.7
IP2N	Untreated	2	0	5	5	5.2 (1.7)	4.2	0.2	0.0	0.1	0.1	0.1	20.8	8.4	29.6
IP3M	Untreated	3	0	68	81	17.7 (1.7)	4.2	2.1	0.0	4.1	0.1	0.2	88.7	7.1	102.3
IP4MF	Untreated	0	0	0	0	9.6 (1.5)	4.4	0.0	2.0	3.1	0.1	0.2	46.3	5.1	56.8
JP11R	LFS-Only	2	1	25	26	6.6 (2.1)	4.6	0.9	0.0	1.8	0.0	0.3	26.6	7.4	37.6
JP2R	LFS-Only	3	2	44	47	3.8 (1.2)	5.2	1.4	0.0	1.5	0.0	0.1	15.6	3.9	22.5
JP4R	LFS-Only	6	1	28	37	6.4 (1.2)	6.5	0.9	0.0	0.7	0.0	0.0	23.3	7.6	32.6
JP5B	LFS-Only	2	0	23	24	13.1 (3.4)	4.5	0.5	0.0	3.1	0.0	0.2	52.1	4.4	60.4
JP5R	LFS-Only	1	0	0	3	4.2 (1.0)	4.5	0.1	0.0	0.3	0.0	0.0	15.7	5.3	21.4
JP7M	LFS-Only	4	0	261	290	18.4 (2.8)	4.9	13.5	6.0	4.3	0.0	0.4	91.3	8.9	124.5
JP9M	LFS-Only	4	0	76	81	18.4 (2.2)	4.6	2.2	0.6	5.7	0.0	0.1	76.4	6.9	92.0
OP1N	Tree-Only	5	70	45	133	17.0 (1.9)	4.3	17.8	2.9	3.2	0.0	0.4	55.9	6.2	86.4
OP2N	Tree-Only	4	41	34	80	10.3 (1.5)	4.2	13.4	0.0	1.9	0.0	0.1	51.7	7.7	75.2
OP3N	Tree-Only	2	117	0	118	10.0 (2.6)	4.4	14.4	0.0	1.5	0.2	0.4	30.0	4.7	51.3
OP4N	Tree-Only	2	79	3	82	11.6 (1.6)	4.5	9.5	0.0	0.6	0.0	0.1	38.3	3.5	52.1
OP5N	Tree-Only	3	89	25	112	13.8 (2.5)	4.5	15.5	10.9	2.7	0.0	0.4	47.6	6.3	83.5
OP6N	Tree-Only	3	72	30	106	11.7 (1.9)	4.5	12.5	0.0	0.6	0.2	0.5	41.9	4.4	60.1
OP7N	Tree-Only	2	87	13	100	17.1 (3.0)	4.4	13.5	0.0	1.9	0.0	0.4	22.5	6.9	45.2
OP8N	Tree-Only	4	71	41	117	12.3 (2.7)	4.3	12.6	3.8	2.1	0.1	0.5	65.5	8.0	92.7
IP1N	LFS-Tree	7	98	167	273	11.7 (1.5)	4.7	27.1	0.8	2.3	0.2	0.1	73.3	8.9	112.7
IP6N	LFS-Tree	6	59	91	168	10.6 (1.9)	5.0	22.8	5.3	0.3	0.1	0.2	57.3	12.9	98.9
JP10R	LFS-Tree	5	57	8	68	9.2 (1.4)	5.1	16.1	3.8	1.1	0.0	0.4	33.4	7.3	62.1
JP1MF	LFS-Tree	5	61	31	101	10.7 (2.2)	5.1	13.9	0.3	1.8	0.0	0.4	47.4	7.6	72.3
JP2N	LFS-Tree	2	74	32	106	8.3 (1.5)	5.0	19.0	0.1	3.4	0.0	0.2	41.0	6.7	70.3
JP3MF	LFS-Tree	3	111	27	139	10.8 (1.8)	5.0	21.5	0.0	0.1	0.0	0.2	52.4	8.4	82.7
JP3N	LFS-Tree	4	115	69	193	24.2 (2.4)	4.5	29.8	2.6	3.5	0.2	0.1	77.8	8.8	122.8
JP4N	LFS-Tree	4	73	51	129	6.1 (2.3)	4.9	21.8	4.0	1.1	0.1	0.1	14.1	13.7	55.0
JP5N	LFS-Tree	6	74	51	141	6.0 (1.2)	4.4	17.5	0.9	0.7	0.1	0.5	31.9	8.7	60.3
JP6N	LFS-Tree	7	56	59	135	7.0 (1.7)	5.4	21.0	2.4	0.9	0.0	0.6	25.1	9.6	59.6
JP8B	LFS-Tree	4	28	120	177	14.5 (3.8)	4.9	11.3	0.0	0.6	0.1	0.5	56.6	6.8	75.9
JP9B	LFS-Tree	7	62	91	167	8.2 (1.8)	4.9	20.4	0.6	4.4	0.1	0.2	41.2	6.8	73.6