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# Soil properties in log landings amended with subsoiling and biochar

**William R. Rumpf**[1] | **Morgan P. Davis**[1] | **John M. Kabrick**[2] | **John Stanovick**[3] | **Lauren S. Pile Knapp**[2] | **Deborah Page-Dumroese**[4]

[1]School of Natural Resources, University of Missouri, Columbia, Missouri, USA

[2]USDA Forest Service, Northern Research Station, Columbia, Missouri, USA

[3]USDA Forest Service, Northern Research Station, Long Pond, Pennsylvania, USA

[4]USDA Forest Service, Rocky Mountain Research Station, Moscow, Idaho, USA

**Correspondence**
William R. Rumpf, School of Natural Resources, University of Missouri, 4500 E Deer Park Rd, Columbia, MO 65201, USA.
Email: wrrwc7@umsystem.edu

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**Abstract**

Soils on log landings can be significantly degraded as a result of harvesting equipment. We evaluated the efficacy of biochar application and subsoiling to reduce compaction and improve soil properties. Fifteen log landings were selected across three US Midwest national forests: the Hoosier National Forest in Indiana, the Mark Twain National Forest in Missouri, and the Shawnee National Forest in Illinois. Landings were divided into four treatments: (1) biochar applied on the soil surface (22.4 Mg ha$^{-1}$), (2) subsoiling (depth of 50 cm), (3) a combination of subsoiling and biochar (22.4 Mg ha$^{-1}$), and (4) a control. Treatments were divided into split-plots and were seeded with a competitive native seed mix or not seeded. Prior to treatment, soil bulk density was assessed (0–30 cm) to determine initial physical properties. Similarly, 1 year following treatments, soil samples were again collected from 0 to 30 cm to assess differences in soil properties among treatments. We found surface applications of biochar reduced bulk density by 18%–19% in the top 10 cm of soil regardless of seeding or when combined with subsoiling ($p < 0.05$). Soils amended with biochar also saw a significant increase in potassium (K), and plant available phosphorus (P), as well as marginal increases in active carbon (C), total C, and nitrogen (N). When compared to pretreatment data, the bulk density decreased significantly ($p = 0.001$) which may be attributed to the establishment vegetation and natural freeze thaw cycles.

**Plain Language Summary**

Log landings are clearings in a forest where trees are sorted and loaded onto trucks after being cut. These clearings could be used to create new habitat for pollinators like bees and butterflies, which are in decline. However, the heavy machinery used on landings often leaves behind compacted soil that makes it hard for plants to regrow. Our study tested whether soil treatments, subsoiling (loosening the soil) and adding

**Abbreviations:** BMP, best management practice; CFs, coarse fragments; ECEC, effective cation exchange capacity; HNF, Hoosier National Forest; MTNF, Mark Twain National Forest; PMC, potentially mineralizable carbon; POXC, permanganate oxidizable carbon; SNF, Shawnee National Forest.

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biochar (a charcoal like material), could improve the soil for plant growth on log landings in three national forests. A year later, we found that biochar reduced soil compaction by up to 19% in the top layer and added helpful nutrients. The treatments reduced soil compaction at shallow depths, but the effect was not seen deep down in the soil or in the other forests. This research shows that with the right treatment, log landings may offer a new opportunity to support pollinator habitat restoration on previously disturbed forest land.

## 1 INTRODUCTION

Log landings are present within most harvest units (Lee et al., 2021) and are cleared areas that allow for staging equipment, for sorting and loading harvested trees for transport during a timber sale (Enyart et al., 2009), and for piling non-merchantable woody residues. They are strategically placed near roads to facilitate harvest removals (Grether & Creagh, 2018) or access to slash piles for burning. They are an important part of the timber harvest infrastructure, relegating heavy machinery traffic to a small area, thereby reducing overall disturbance of the wider forest (Grether & Creagh, 2018). To minimize their areal extent, landings are often reused for successive timber harvest entries (Grether & Creagh, 2018). While landings can be beneficial to overall forest health and soil productivity by reducing widespread impacts from harvesting, heavy traffic concentrated in a small area can lead to significant soil degradation (Hatchell et al., 1970). Compaction by heavy machinery reduces macroporosity (Williamson & Neilsen, 2000), which subsequently decreases water and gas movement through the soil, infiltration, water holding capacity, and gas exchange and can also increase runoff potential (Kozlowski, 1999; Sidle, 1980; Zenner et al., 2007). Increased soil bulk density associated with compaction restricts root growth, which can slow or eliminate revegetation along with increasing risks of erosion and water resource sedimentation (Kozlowski, 1999). Another potential side effect of concentrating wood and residues at landings is the accumulation slash mats (Han et al., 2009). These thick layers of bark, mulch, or slash can bury mineral soil limiting seed germination, and in some circumstances immobilize plant available nitrogen, reducing fertility (Homyak et al., 2008).

Current best management practices (BMPs) for log landings in the US Midwest focus on stabilizing the soil after timber harvesting is completed. Recommendations include applying mulch residues, diverting water by constructing water bars on skid trails leading to and from landings, and seeding with noninvasive annual plant species (Enyart et al., 2009; Grether & Creagh, 2018). Noninvasive crop seed such as annual rye, oats, or soybeans are commonly used to provide rapid revegetation (Binkley & Fisher, 2019). Postharvest log landing BMPs seldom include decompaction, opting instead to avoid compaction by adhering to soil protection methods (Cambi et al., 2015). These include careful planning of log landing placement and restricting harvest operations when soils are close to saturation and most vulnerable to compaction (Cambi et al., 2015; Page-Dumroese et al., 2006). Compaction prevention strategies such as the use of slash mats (Han et al., 2009) or wide, low-pressure tires (Cambi et al., 2015) are often recommended for vulnerable soils to prevent severe compaction. Log landings are often reused in future harvests, are seldom planted with native tree species, and are generally not managed as high-quality wildlife habitat (Enyart et al., 2009; Grether & Creagh, 2018). However, there are potential opportunities to create wildlife habitat in between harvest operations (Grether & Creagh, 2018; Jackson et al., 2014; Tucker & Olson, 1994). Log landings, as forest openings, can increase floral and nesting resources to support pollinator populations within forests (Lee et al., 2021). As active forest management on public lands is often designed to restore, sustain, and promote resilience of forest communities, log landings, as the byproduct of timber operations, may help to enrich pollinator habitat but soil conditions may not be favorable to pollinator-friendly, diverse native plants.

To improve log landings for pollinator habitat, soil compaction and fertility issues may need to be addressed to promote establishment of vegetation for nesting and floral resources. Non-inversion tillage practices and biochar additions are two management strategies that lend well to the overall reduction of soil compaction and improvement of soil properties for successful vegetation establishment.

Traditional forms of tillage such as rototilling, while useful in restoring unobstructed compacted soil, can be difficult to implement in log landing soils where obstacles such as stumps, roots, buried logs, and coarse fragments (CFs) or rocks are abundant (Andrus & Froehlich, 1983). Subsoiling has shown promise as an alternative form of tillage to reduce soil compaction (McNabb, 1994; Plotnikoff et al., 2002) and can be used in soils with these obstructions (Andrus & Froehlich, 1983). Unlike rototilling, subsoiling breaks and lifts soil with vertical tines allowing the soil to maintain or regain its natural structure and horizonation.

Biochar is a carbon (C)-rich material created through pyrolysis, heating biomass at high temperatures in the absence of oxygen. Pyrolysis causes the thermal decomposition of the original biomass feedstock and results in a stable form of carbon that can persist in soils for hundreds to thousands of years (Verheijen et al., 2010). Biochar's stability has garnered significant attention in the past decade as a potential tool for long term C storage and improvement of soil physiochemical properties (Thomas & Gale, 2015). When added to soil, biochar has been shown to decrease soil bulk density by 31% and increase porosity by 64%, depending on soil type and texture (Blanco-Canqui, 2017). This increase in porosity can improve infiltration and water storage properties (Novak et al., 2016), leading to lower bulk density and better establishment of vegetation. Biochar is generally ∼80% carbon and can have some nutrients, but the quantity and availability of these depends on feedstock type, pyrolysis temperature, and residence time in the reactor. Efforts to use non-merchantable woody residues to create biochar are growing. Work in the western United States indicates that biochar is a good soil amendment to increase soil C, microbial activity, or restore soil physical properties while also reducing the risk of wildfire by converting potential fuel sources to biochar (Page-Dumroese et al., 2024). In addition, biochar can be used to remediate mine soils (Rodriguez Franco & Page-Dumroese, 2021), be a climate-smart forestry tool (Rodriguez Franco et al., 2024), and used in forest seedling production (Dumroese et al., 2020).

As part of a larger project to promote the establishment of native pollinator habitat on log landings, subsoiling and biochar treatments were installed on 15 log landings across Mark Twain National Forest, Shawnee National Forest, and Hoosier National Forest (HNF) in the US Midwest. The objectives of this study were to measure the impacts of these two treatments on soil physical, chemical, and biological soil properties across a broad range of log landings varying in intensity of use, landscape position, and soil texture. We hypothesized subsoiling treatments would decrease bulk density, and that additions of biochar would decrease bulk density and increase soil carbon.

**Core Ideas**

- Biochar application reduced bulk density and improved soil properties.
- Subsoiling had little effect on soil properties.
- There were no significant interactions between biochar and subsoiling.
- Biochar significantly increased available K and P.

## 2 | MATERIALS AND METHODS

### 2.1 | Study sites

On the HNF in Indiana, the Mark Twain National Forest (MTNF) in Missouri, and in the Shawnee National Forest (SNF) in Illinois, five log landings in each forest were randomly selected ($n$ = 15 landings) ranging in size from 0.01 to 0.35 ha. Soils in the landings covered a range of properties according to USDA-NRCS maps (Table 1). The soils in the log landings on the HNF and SNF were mapped as having fine or fine-silty particle size class distributions and lacked substantial CFs in surface horizons. Soils among the log landings on the MTNF were more variable, and included soils with fine, loamy-skeletal, and loamy-skeletal over clayey particle size class distributions with varying amounts of CFs in surface horizons (USDA-NRCS, 2025). (Table 1).

### 2.2 | Study design and treatments

A randomized complete block split-plot design was implemented on each landing. The landings served as statistical blocks and each of the 15 landings were divided into four plots to test whole-plot treatments of biochar added at 22.4 Mg ha$^{-1}$, subsoiling to a depth of 50 cm, biochar (22.4 Mg ha$^{-1}$) application followed by subsoiling treatment, and a control (Figure S1). The choice of application rate was informed by Rodriguez Franco and Page-Dumroese (2021), which saw improvements in chemical and physical properties at this rate. Each whole plot treatment was divided into two equal subplots and randomly assigned a seeding treatment (seeded, not seeded). A single source softwood biochar derived from loblolly pine (Wakefield Biochar) was applied using an industrial self-propelled fertilizer broadcaster at a rate of 22.4 Mg ha$^{-1}$ to the surface of the mineral soil of the biochar amendment plots. Biochar chemical composition and characterization can be found in the supplemental material (Table S1). Subsoiling was implemented using a skid steer equipped with two, 50-cm-long dual directional ripping teeth spaced 60-cm apart. The teeth were anchored to a CLFab XR Ripper equipped with gauge wheels to ensure consistent depth. Tines were pulled through each subplot in reverse to prevent the skid steer from driving over treated soil. A combination treatment of biochar application at 22.4 Mg ha$^{-1}$ was followed by the same subsoiling treatment to a depth of 50 cm. The order of these treatments was maintained to incorporate the biochar on the surface of the soil.

The seeded treatment received a native seed mix of herbaceous species representing 28 forbs and four grasses. The native seed mix was designed to provide habitat for pollinator

**TABLE 1** Landing information for treated landings in Hoosier National Forest (HNF), Mark Twain National Forest (MTNF), and Shawnee National Forest (SNF). Coordinates, soil series/family, mean coarse fragments in % volume from pretreatment data, date of last harvest-related activity, and climate information. Soil Series and Family (USDA-NRCS, 2025) and Climate Data (National Climatic Data Center, 1981–2020 climate norms).

| Forest | Landing ID | Location | Surrounding soil series and family | Date of last activity | Mean annual temperature (°C) | Mean annual precipitation (cm) |
|---|---|---|---|---|---|---|
| Hoosier NF | HNF 2 | 38.162002299758626, −86.47752183427556 | Apalona-Zanesville: Fine-silty, mixed, active, mesic Oxyaquic Fragiudalfs | 2019–2020 | 13.17 | 130.25 |
| | HNF 3 | 38.16571765797519, −86.48323157157961 | Apalona-Zanesville: Fine-silty, mixed, active, mesic Oxyaquic Fragiudalfs | 2019–2020 | 13.17 | 130.25 |
| | HNF 6 | 38.29670041790447, −86.6486681680185 | Cuba: Fine-silty, mixed, active, mesic Fluventic Dystrudepts | 2019–2020 | 13.17 | 130.25 |
| | HNF 9 | 38.32184583762452, −86.6627033175201 | Apalona-Zanesville: Fine-silty, mixed, active, mesic Oxyaquic Fragiudalfs | 2019–2020 | 13.17 | 130.25 |
| | HNF 11 | 38.21969941808925, −86.63834256599426 | Ebal-Deuchars-Kitterman: Fine, mixed, active, mesic Oxyaquic Hapludalfs | 2019–2020 | 13.17 | 130.25 |
| Mark Twain NF | MTNF 3 | 36.98536870665993, −90.47660523670386 | Poynor-Clarksville-Scholten: Loamy-skeletal over clayey, siliceous, semiactive, mesic Typic Paleudults | 2019–2020 | 14.83 | 132.69 |
| | MTNF 7 | 36.98709887231421, −90.47132557138178 | Poynor-Clarksville-Scholten: Loamy-skeletal over clayey, siliceous, semiactive, mesic Typic Paleudults | 2019–2020 | 14.83 | 132.69 |
| | MTNF 12 | 36.895406575069856, −91.2126242573373 | Clarksville: Loamy-skeletal, siliceous, semiactive, mesic Typic Paleudults | 2019–2020 | 12.22 | 128.37 |
| | MTNF 15 | 37.483066626114876, −92.01934837130462 | Viburnum: Fine, mixed, active, mesic Aquic Paleudults | 2019–2020 | 12.94 | 118.21 |
| | MTNF 16 | 37.48046576873601, −92.02023786774372 | Bendavis-Poynor: Loamy-skeletal, siliceous, active, mesic Typic Hapludults | 2019–2020 | 12.94 | 118.21 |
| Shawnee NF | SNF 3 | 37.55122565653777, −88.27710847974659 | Grantsburg: Fine-silty, mixed, active, mesic Oxyaquic Fragiudalfs | 2021 | 12.67 | 129.26 |
| | SNF 6 | 37.55587866610417, −88.27084704141063 | Zanesville: Fine-silty, mixed, active, mesic Oxyaquic Fragiudalfs | 2021 | 12.67 | 129.26 |
| | SNF 8 | 37.54691554306477, −88.28945660604421 | Grantsburg: Fine-silty, mixed, active, mesic Oxyaquic Fragiudalfs | 2021 | 12.67 | 129.26 |
| | SNF 9 | 37.54899576327478, −88.28726792582853 | Grantsburg: Fine-silty, mixed, active, mesic Oxyaquic Fragiudalfs | 2021 | 12.67 | 129.26 |
| | SNF 10 | 37.55257790068138, −88.2882298363115 | Grantsburg: Fine-silty, mixed, active, mesic Oxyaquic Fragiudalfs | 2021 | 12.67 | 129.26 |

species. A light application of straw was applied by hand to seeded subplots after hand broadcasting the seed mix.

Soil treatments were implemented in March–April of 2021. Control plots were designed to match standard log landing retirement practices. However, initial log landing conditions varied by forest and by landing. The accumulation of organic debris was extensive on some landings of MTNF and HNF, where slash mats occasionally reached depths of 30+ cm. On these forests, the slash mats were removed by a skid steer prior to the implementation of the study treatments to maintain consistency. Care was taken to only strip off residue to the surface of the mineral soil. In addition, some HNF landings had an additional topsoil salvage practice in which the topsoil was removed prior to harvest and returned postharvest to reduce erosion.

## 2.3 | Sample collection

Soil sampling locations within each treatment plot were generated at random for each sampling period using ArcGIS (ESRI). One sample was taken from each subplot before and after treatment implementation. This includes seeded and unseeded subplots for a total of 360 samples across three forests, five landings per forest, eight subplots per landing, and three depths per subplot. Pretreatment sampling was conducted from January to March 2021 and posttreatment sampling was done approximately 1 year later, from February to April 2022. At each sampling location, a 40-cm deep hole was dug using a tile spade, without compressing the sampling surface on the inside of the hole. A 5-cm tall, beveled metal ring with a 7.2-cm inner diameter (203.57 cm$^3$) was pushed into a side of the hole at the approximate center of three depth intervals from the surface of the mineral soil: 0–10 cm, 10–20 cm, and 20–30 cm. Each sample was then excavated from the side of the hole and trimmed to the extent of the metal ring. The samples were then transferred to plastic bags and placed in a cooler for transport. Samples were stored in a 4˚C refrigerator prior to sample processing. Soil samples were air dried for a minimum of 1 week, until a consistent total weight was reached. Pretreatment samples were only analyzed for bulk density, while posttreatment samples were analyzed for a suite of soil analyses.

## 2.4 | Soil measurements

### 2.4.1 | Bulk density

Soil compaction was measured through fine earth fraction bulk density ($\rho$bs) via ring sampling. Bulk density was assessed pretreatment and 1-year posttreatment. Each sample was disaggregated and homogenized with the CFs removed. CFs were sorted by type (wood or rock) and then washed, oven dried and massed. Mass and volume of CFs were measured and calculated using the generalized densities of rock at 2.65 g cm$^{-3}$ and shortleaf pine bark chips/woodchips at 0.526 g cm$^{-3}$ (Miles & Smith, 2009). These masses and volumes were subtracted from the overall sample to calculate density of the fine earth fraction (Andraski, 1991).

Volume of rocky CFs:

$$v_r = \rho_{bT}\left(\frac{g_r}{\rho_{br}}\right)$$

Volume of woody CF:

$$v_w = \rho_{bT}\left(\frac{g_w}{\rho_{bw}}\right)$$

Fine earth fraction bulk density:

$$\rho_{bS} = \rho_{bT}\left(\frac{1-g_r-g_w}{1-v_r-v_w}\right)$$

where:

– $\rho_{bT} =$ total bulk density of sample in g cm$^{-3}$ with CFs included.

– $\rho_{bS} =$ bulk density of *fine earth fraction (CFs removed)* in g cm$^{-3}$

– $v_r =$ volume of rocky CFs volume in cm$^3$

– $g_r =$ rocky CFs mass in grams

– $v_w =$ volume of woody CFs volume in cm$^3$

– $g_w =$ woody CFs mass in grams

### 2.4.2 | Chemical and biological properties

Soil chemical and biological properties were compared between soil treatments 1-year following treatment. Soil carbon was determined through active carbon and total carbon. Active carbon was determined via a permanganate oxidizable carbon (POXC) method originally described in Coleman (2012) and Hedges et al. (1999). An average between standard duplicate sets was taken to create a standard curve to interpolate the concentration of POXC present in the samples. The samples were run on a BioTek Epoch microplate spectrophotometer (Bio Tek Instruments). POXC calculations were derived from Weil et al. (2003):

$$\frac{\text{mg POXC}}{\text{kg soil}} = \left[0.02\frac{\text{mol}}{\text{L}} - (a + b \times \text{Abs})\right] \times (9000\frac{\text{mgC}}{\text{mol}} \times \frac{0.02\text{L rx solution}}{\text{soil weight}}$$

Percent total C and N were measured on a Leco TruSpec CN (Leco Corp.) through dry combustion at 950˚C. Air-dried subsamples were ground and homogenized before being measured and rolled into foil sheets for the Leco. Phosphorus

(P) was determined via Bray P-1 extraction (Bray & Kurtz, 1945) and measured on a Milton Roy Spectronic 20 D+ spectrophotometer (Water Corporation). Exchangeable base cations (calcium [Ca], magnesium [Mg], sodium [Na], and potassium [K]) were determined simultaneously with effective cation exchange capacity (ECEC) via $NH_4Cl$ extraction followed by Atomic Emission Spectrometry. ECEC was measured after $NH_4Cl$ extraction followed by Kjeldahl titration on a Foss Kjeltec 8200 Tecator (Foss A/S). Exchangeable cations (Ca, Mg, Na, and K) were measured using atomic emission spectroscopy on an Agilent 4210 MP-AES (Agilent Technologies). Soil pH was measured using a Fisher Scientific Accumet XL600 pH electrode probe in a 2:1 by mass water to soil slurry.

Potentially mineralizable C (PMC), the only biologically categorized measurement, was determined via $CO_2$ burst method (Franzluebbers, 2018) at 50% water filled pore space within modified 125-mL Erlenmeyer flasks over a 24-h period using a Licor Li-830 $CO_2$ Gas Analyzer (Li-Cor Biosciences).

### 2.5 | Statistical analysis

Bulk density was the only analysis that was completed for pre- and posttreatment samples due to budget constraints. To examine the effect of treatment implantation in bulk density, a paired *t*-test was conducted for each treatment at each depth. Treatment effects on posttreatment soil measurements were analyzed using a generalized linear mixed model with PROC GLIMMIX in SAS 9.4 (SAS Institute, Inc.). There were 12 response variables and three soil depths for a total of 36 GLIMMIX Procedure models. Log landing within forests was included as a random effect and soil treatments were included as a fixed effect. The Kenward-Rogers denominator degrees of freedom adjustment method (Littell et al., 2006) was used in all models. The GLMMIX procedure was chosen for its ability to fit statistical models to non-normally distributed data and to accommodate noncontinuous response variables and nonconstant variance. If the treatment effect was significant ($p < 0.05$), a Tukey–Kramer post hoc test was used to determine significance between individual treatment means. Only main effects are presented here because the seeding treatment (split-plot) was not significant across any analysis.

## 3 | RESULTS AND DISCUSSION

### 3.1 | Bulk density

Pretreatment fine earth fraction bulk density data varied between forests and generally increased with depth from 0 to 30 cm. HNF had the highest average bulk densities at 1.43 g $cm^{-3}$ at 0–10 cm, 1.5 g $cm^{-3}$ at 10–20 cm, and 1.52 g $cm^{-3}$ at 20–30 cm. MTNF had the lowest average pretreatment densities, measuring 1.11 g $cm^{-3}$ at 0–10 cm, 1.27 g $cm^{-3}$ at 10–20 cm, and 1.23 g $cm^{-3}$ at 20–30 cm. Average SNF densities were measured at 1.31, 1.42, and 1.41 g $cm^{-3}$ at depths of 0–10 cm, 10–20 cm, and 20–30 cm, respectively. Notable change in bulk density was observed from 0 to 10 cm across all forests (Figure 1). Posttreatment fine earth fraction bulk density in the top 0–10 cm of soil across all treatments and forests ranged from 0.22 to 1.67 g $cm^{-3}$ with an overall average of 1.01 g $cm^{-3}$, compared to a pretreatment average of 1.3 g $cm^{-3}$. Individual comparisons of plots pre- and posttreatment implementation at 0–10 cm yielded *p*-values less than 0.001, signifying change across treatments including the control plots. The change across treatments in the control plot is likely due to the removal of woody debris and the establishment of volunteer vegetation. Change in bulk density was not observed at 10- to 20-cm or 20- to 30-cm depths ($p < 0.05$).

Within treatments the lowest mean bulk density occurred with the combination subsoiling and biochar treatment (B × S) at 0.92 g $cm^{-3}$ followed by biochar amendment at 0.93 g $cm^{-3}$, subsoiling at 1.07 g $cm^{-3}$, and the control treatment at 1.14 g $cm^{-3}$. Compared to control, biochar amendment and B × S were marginally significantly less ($p = 0.065$) at the 0- to 10-cm depth (Figure 2). Bulk density increased with depth, ranging from 0.47 to 2.12 g $cm^{-3}$, and with a mean of 1.42 g $cm^{-3}$ at the 10- to 20-cm depth. At the 10- to 20-cm depth, bulk density of the biochar treatment was significantly less ($p = 0.015$) compared to control. At the 20- to 30-cm depth bulk density ranged from 0.26 to 1.87 g $cm^{-3}$ with a mean of 1.36 g $cm^{-3}$, and no significant differences ($p > 0.050$) were found among treatments (Table 2).

### 3.2 | Cations, pH, and phosphorus

Posttreatment differences in cations, pH, and phosphorus varied among soil treatment plots. There were no significant differences between treatments at any depth ($p > 0.05$) for ECEC. In upper 0- to 10-cm depth, ECEC ranged from 3.88 cmolc $kg^{-1}$ soil to 32.28 cmolc $kg^{-1}$ with an average of 12.27 cmolc $kg^{-1}$. In the 10- to 20-cm depth, ECEC ranged from 1.94 to 25.7 cmolc $kg^{-1}$ with an average of 9.85 cmolc $kg^{-1}$. In the bottom 20- to 30-cm depth, ECEC ranged of 2.77–23.3 cmolc $kg^{-1}$ with an average of 11.14 cmolc $kg^{-1}$. Exchangeable base cations, however, differed among treatments. Potassium was significantly different among treatments ($p = 0.051$), with significant increases in K in the biochar ($p = 0.017$) treatment compared to control and subsoiling (Figure 3). Magnesium was not significant at any depth among treatments ($p = 0.148$). Unlike some of the exchangeable cations, soil pH ranged from 4.2 to 7.8, with an overall average of 5.6 across all depths. There were no significant differences between treatments ($p > 0.050$), with a trend of

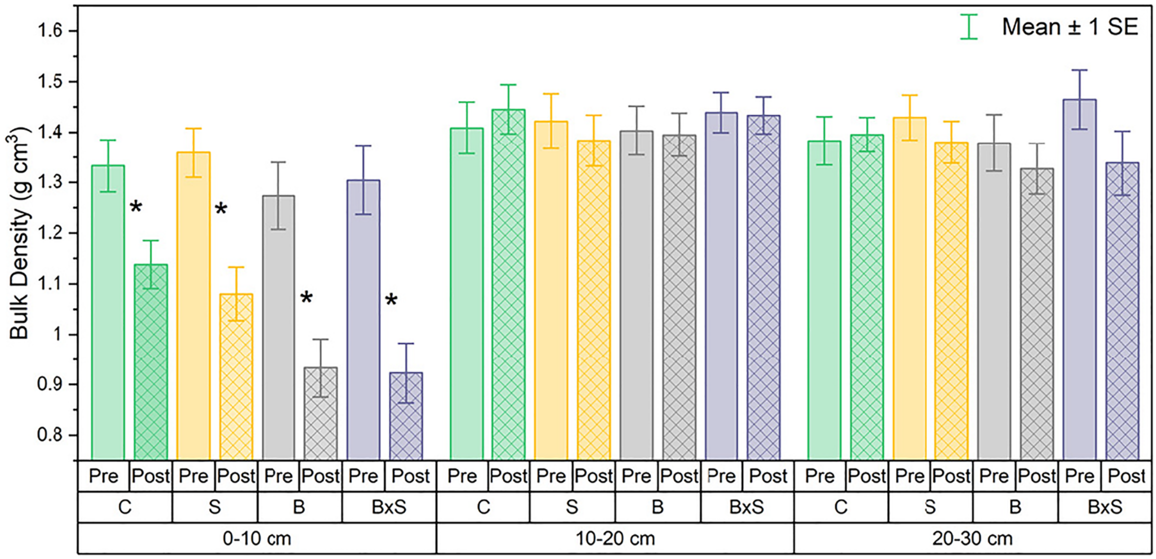


**FIGURE 1** Mean bulk densities presented with standard error for pretreatment and posttreatment plots. Treatments and their notation are control (C, green), subsoiling (S, yellow), biochar (B, gray), and biochar and subsoiling combination (B × S, purple). Data labeled pre were samples collected prior to treatment implementation. Data labeled post were collected 1 year after treatment implementation. Asterisk denotes significance difference between pre- and posttreatment at $p < 0.001$.

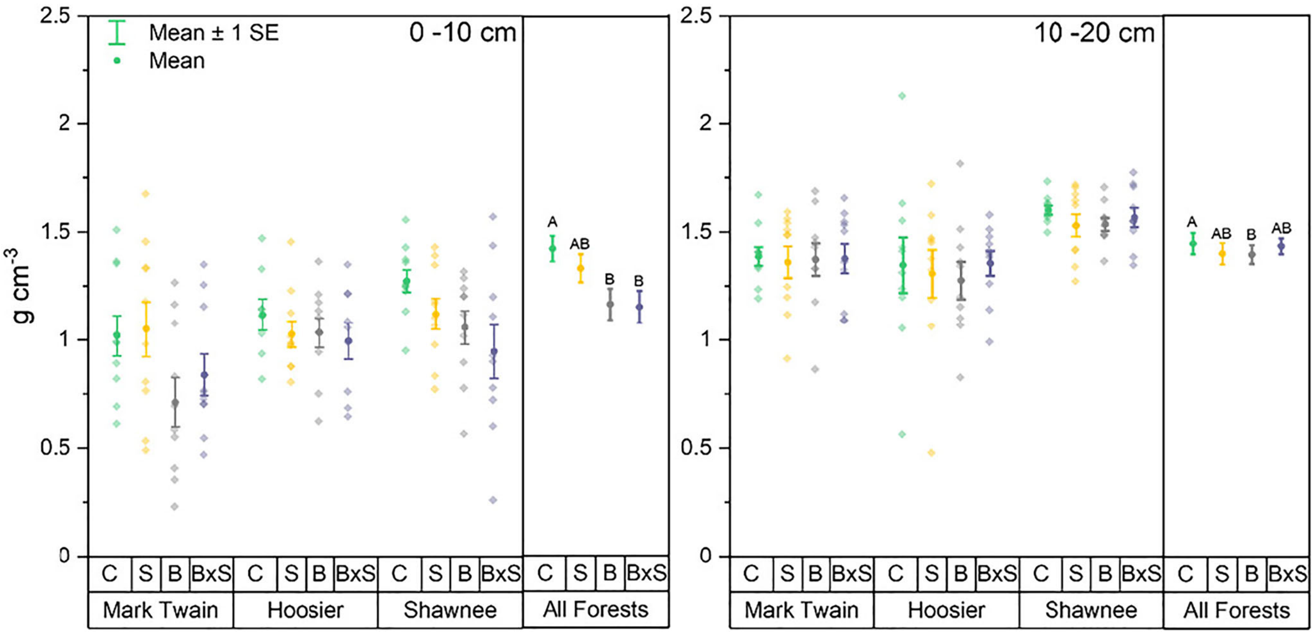


**FIGURE 2** Fine earth fraction bulk densities at the 0- to 10- and 10- to 20-cm depth by forest and treatment, with the post-analysis results combined. Capital letters above means indicate significant differences among treatments ($p < 0.05$). Lowercase letters above means indicate marginal significance ($p < 0.1$). Treatments and their notation are control (C, green), subsoiling (S, yellow), biochar (B, gray), and biochar and subsoiling combination (B × S, purple).

increasing acidity moving down the soil profile, from 6.0 at 0–10 cm to 5.3 at 20–30 cm.

Bray P-1 ranged from 0.4 to 75.4 mg kg$^{-1}$ with an average of 6.4 mg kg$^{-1}$ for the 0- to 10-cm depth. There was a significant increase in P with biochar additions at the 0- to 10-cm depth with significantly greater P concentrations in the biochar treatment ($p = 0.001$) and B × S ($p < 0.001$) compared to control (Figure 3). Phosphorus concentration decreased with depth and ranged from 0.1 to 19.7 mg kg$^{-1}$, with an average of 3.26 mg kg$^{-1}$ at the 10- to 20-cm depth, and from 0.1 to 15.8 mg kg$^{-1}$, with an average of 2.5 at the 20- to 30-cm depth, with no significant differences in treatments at these two.

**TABLE 2** Analysis of variance (ANOVA) results showing LS means and standard error in parentheses.

| | | Treatments | | | | | |
|---|---|---|---|---|---|---|---|
| **Properties** | **Depth (cm)** | **Biochar** | **Control** | **Subsoiling** | **Biochar + subsoiling** | ***F* value** | **Pr > *F*** |
| Bulk density ($g/cm^3$) | 0–10* | 0.93 (0.07) | 1.14 (0.07) | 1.07 (0.07) | 0.92 (0.07) | 4.15 | 0.0654 |
| | 10–20* | 1.39 (0.07) | 1.45 (0.07) | 1.4 (0.07) | 1.43 (0.07) | 5.44 | 0.0379 |
| | 20–30* | 1.33 (0.09) | 1.4 (0.09) | 1.38 (0.09) | 1.34 (0.09) | 1.28 | 0.3641 |
| Total carbon (%) | 0–10 | 4.15 (1.2) | 2.57 (1.21) | 2.5 (1.2) | 4.12 (1.21) | 2.88 | 0.1226 |
| | 10–20 | 0.88 (1.19) | 0.75 (1.19) | 0.84 (1.19) | 0.74 (1.19) | 0.72 | 0.5736 |
| | 20–30 | 0.67 (1.14) | 0.55 (1.14) | 0.58 (1.14) | 0.5 (1.14) | 1.46 | 0.3163 |
| Total nitrogen (%) | 0–10 | 0.18 (1.13) | 0.13 (1.14) | 0.11 (1.13) | 0.15 (1.14) | 2.4 | 0.1497 |
| | 10–20 | 0.06 (1.26) | 0.06 (1.26) | 0.06 (1.26) | 0.05 (1.26) | 0.33 | 0.8071 |
| | 20–30 | 0.05 (1.17) | 0.04 (1.17) | 0.04 (1.17) | 0.04 (1.17) | 0.69 | 0.5884 |
| pH | 0–10 | 5.89 (1.03) | 5.79 (1.03) | 5.96 (1.03) | 6.05 (1.03) | 1.12 | 0.4107 |
| | 10–20 | 5.44 (1.03) | 5.49 (1.03) | 5.56 (1.03) | 5.49 (1.03) | 0.24 | 0.8649 |
| | 20–30 | 5.16 (1.03) | 5.33 (1.03) | 5.41 (1.03) | 5.12 (1.03) | 0.87 | 0.5071 |
| POXC (mg/kg) | 0–10* | 843.11 (62.11) | 656.45 (64.55) | 661.74 (62.11) | 735.84 (63.26) | 2.08 | 0.1093 |
| | 10–20 | 236.42 (1.98) | 97.65 (1.98) | 145.39 (1.98) | 131.51 (1.98) | 0.89 | 0.4478 |
| | 20–30* | 221.34 (1.26) | 195.66 (1.26) | 247.7 (1.26) | 190.01 (1.26) | 0.78 | 0.5467 |
| PMC (mg/kg) | 0–10* | 156 (18.6) | 115.6 (18.98) | 124.54 (18.29) | 155.7 (19.3) | 1.44 | 0.3272 |
| | 10–20 | 14.34 (1.31) | 13.7 (1.31) | 15.73 (1.31) | 12.16 (1.31) | 0.5 | 0.6936 |
| | 20–30 | 8.44 (1.13) | 9.31 (1.18) | 12.4 (1.2) | 8.38 (1.13) | 1.85 | 0.1504 |
| ECEC (cmol/kg) | 0–10 | 11.94 (1.13) | 10.61 (1.13) | 11.03 (1.13) | 11.63 (1.13) | 0.43 | 0.7415 |
| | 10–20 | 8.08 (1.24) | 9.2 (1.24) | 8.48 (1.24) | 9.61 (1.24) | 0.98 | 0.4616 |
| | 20–30 | 8.91 (1.2) | 10.53 (1.2) | 10.11 (1.2) | 10.8 (1.2) | 1.37 | 0.3376 |
| Bray 1-P (mg/kg) | 0–10 | 5 (1.44) | 3.02 (1.44) | 3.51 (1.44) | 5.75 (1.44) | 7.65 | 0.0002 |
| | 10–20 | 2.2 (1.6) | 1.85 (1.6) | 1.97 (1.61) | 2.06 (1.6) | 0.14 | 0.9355 |
| | 20–30 | 1.19 (1.69) | 1.21 (1.69) | 1.36 (1.69) | 1.16 (1.69) | 0.1 | 0.9542 |
| Mg (cmolc/kg) | 0–10 | 1.74 (1.16) | 1.28 (1.16) | 1.39 (1.16) | 1.58 (1.16) | 1.82 | 0.1485 |
| | 10–20 | 0.92 (1.4) | 1.06 (1.4) | 0.92 (1.4) | 1.11 (1.4) | 0.27 | 0.8451 |
| | 20–30* | 0.98 (1.33) | 1.51 (1.33) | 1.26 (1.32) | 1.42 (1.33) | 1.62 | 0.2802 |
| Ca (cmolc/kg) | 0–10 | 6.24 (1.29) | 4.14 (1.29) | 4.68 (1.29) | 5.99 (1.29) | 1.23 | 0.3789 |
| | 10–20 | 1.81 (1.75) | 1.65 (1.75) | 1.59 (1.76) | 1.8 (1.75) | 0.09 | 0.9606 |
| | 20–30 | 1.41 (1.45) | 1.65 (1.45) | 1.58 (1.45) | 1.4 (1.46) | 0.2 | 0.895 |
| Na (cmolc/kg) | 0–10 | 0.16 (1.61) | 0.15 (1.61) | 0.18 (1.61) | 0.16 (1.61) | 0.89 | 0.4576 |
| | 10–20 | 0.18 (1.61) | 0.16 (1.63) | 0.18 (1.62) | 0.16 (1.61) | 0.11 | 0.9469 |
| | 20–30 | 0.16 (1.36) | 0.14 (1.37) | 0.16 (1.36) | 0.12 (1.36) | 1.17 | 0.333 |
| K (cmolc/kg) | 0–10 | 0.27 (1.48) | 0.19 (1.48) | 0.19 (1.48) | 0.22 (1.48) | 2.69 | 0.0512 |
| | 10–20 | 0.14 (1.35) | 0.12 (1.35) | 0.12 (1.35) | 0.14 (1.35) | 1.89 | 0.2294 |
| | 20–30 | 0.14 (1.37) | 0.12 (1.37) | 0.12 (1.37) | 0.15 (1.37) | 0.76 | 0.5575 |

*Note*: Degrees of freedom = 2.
Abbreviations: ECEC, effective cation exchange capacity; PMC, potentially mineralizable carbon; POXC, permanganate oxidizable carbon.
*denotes normal distribution. No superscript indicates log normal distributions.

## 3.3 | Soil carbon

Post treatment total C in the 0 to 10-cm depth ranged from 4.2 to 366.5 g $kg^{-1}$ with an average of 44.7 g $kg^{-1}$. There was marginal significance ($p$ = 0.123) between treatments with the combination treatment B × S and biochar having greater C than control and subsoiling alone (Figure 3). Most of the samples had less than 80 g $kg^{-1}$ C. Five samples had greater than 100 g $kg^{-1}$ carbon and were all from biochar and B × S treatments. Total carbon at the 10- to 20-cm depth ranged

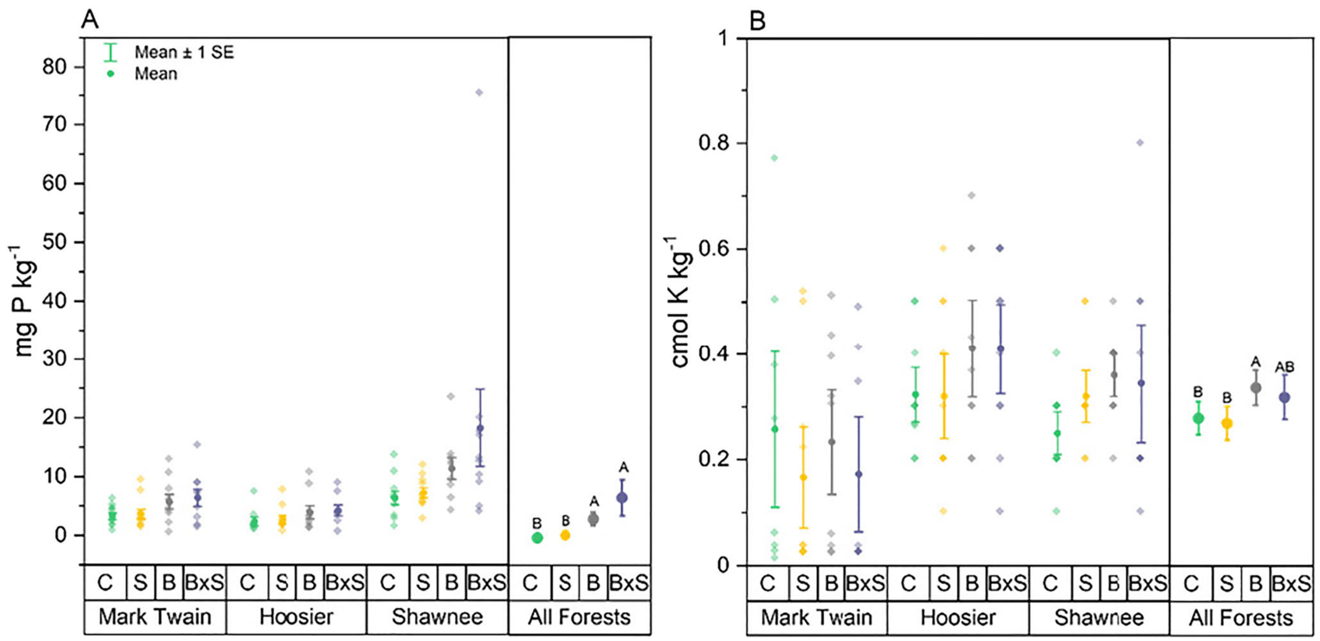


**FIGURE 3** Bray P-1 at the 0- to 10-cm depth (panel A) by forest and treatment, with post-analysis results on the right panel. Letters indicate significance ($p < 0.05$). Exchangeable potassium concentration soil at the 0- to 10-cm depth (panel B) by forest and treatment, with the post-analysis results combined on the right panel. Different uppercase letters indicate significant effects ($p < 0.05$). Treatments are control (C, green), subsoiling (S, yellow), biochar amendment (B, gray), and biochar and subsoiling combination (B × S, purple).

from 2.2 to 43.1 g kg$^{-1}$ with an average of 9.8 g kg$^{-1}$ and no significant difference among treatments. At the 20- to 30-cm depth, values ranged from 1.6 to 64.7 g kg$^{-1}$ and an average of 7.7 g kg$^{-1}$, again with no significant differences between treatments ($p > 0.05$).

Similar to total carbon, active, or labile carbon fraction differences were marginally significant ($p = 0.110$) among treatments in the 0- to 10-cm depth. POXC measurements ranged from 53.1 mg POXC kg$^{-1}$ soil, a maximum of 1448.6 mg POXC kg$^{-1}$ soil. Samples collected from 0–10 cm average of 724.3 mg POXC kg$^{-1}$. There was a significant increase in POXC in the biochar treatments when compared to the control ($p = 0.033$) and subsoil treatments ($p = 0.035$) (Figure 4). In the 10- to 20-cm depth the POXC ranged from 0 mg kg$^{-1}$ soil to 1238 mg kg$^{-1}$, with an average of 300 mg kg$^{-1}$. In the 20- to 30-cm depth, POXC ranged from 0 to 1183.0 mg kg$^{-1}$, with an average of 258.8 mg kg$^{-1}$.

PMC ranged from 4 to 414 mg kg$^{-1}$ with an average of 58 mg kg$^{-1}$ soil across all treatments and depths. There was no significant difference between treatments at any depths ($p > 0.05$); however, there was a trend of increased PMC in the biochar amended soils and the combination treatment B × S in the 0–10 cm (Figure 4). At all depths, there were no significant increases ($p = 0.14$) in total N. Total N ranged from 0.3 to 4.8 g kg$^{-1}$ with an average of 4.8 g kg$^{-1}$ from 0 to 10 cm, 0.5 to 2.0 g kg$^{-1}$ with an average of 0.6 g kg$^{-1}$ from 10 to 20 cm, and 0.02 to 3.1 g kg$^{-1}$ with an average of 0.5 g kg$^{-1}$ from 20–30 cm.

## 4 | DISCUSSION

This study builds on previous forest soil studies to show how impacted log landings can be remediated by the use of biochar and/or subsoiling, which would likely improve these ephemeral sites for pollinator habitat. Because biochar can be produced locally with woody residues from timber harvests (Wilson et al., 2024), there is increasing interest in also using it locally as a soil amendment compacted or otherwise disturbed during timber harvesting. Biochar created from slash and harvest residue reduces fuel sources that could increase fire risk (Wilson et al., 2024) and avoids disposal methods such as pile burning that have negative effects on gas emissions and damage to soils (Puettmann et al., 2020; Wilson et al., 2024). Log landings have also been identified as significant methane sources (Keller et al., 2005; Vantellingen & Thomas, 2021), and biochar has shown promise in reducing these effects (Thomas et al., 2022) Biochar, as a low density, high porosity material, can reduce bulk density while also increasing CEC, water holding capacity (Blanco-Canqui, 2021), and available nutrients (Lehmann & Joseph, 2015). Due to its resistance to decomposition, biochar tends to maintain its form across longer periods when compared to the woody residue that is often left on landings. This recalcitrance combined with the high aspect ratio typical of wood-derived biochars, allows it to augment soil pore space for long periods of time. This inherent pore space as well as formation of biochar–soil interpores and increased aggregation has the

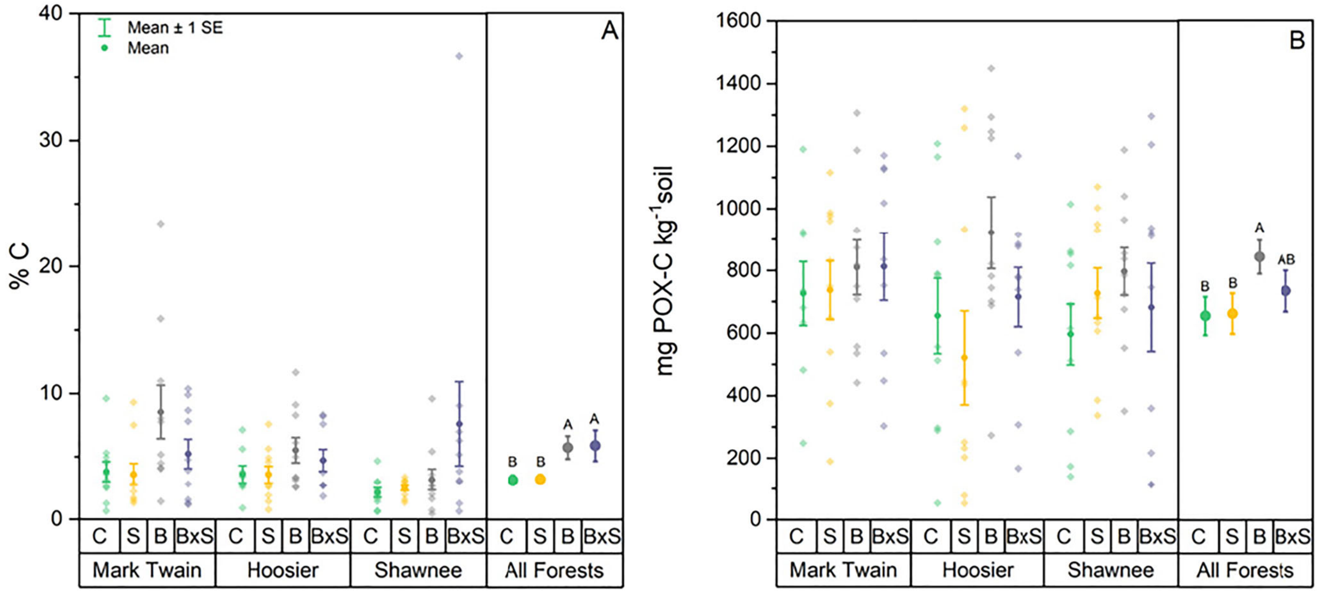


**FIGURE 4** Total carbon at the 0- to 10-cm depth (panel A) by forest and treatment, with post-analysis results on the right panel. Lowercase letters indicate marginal significance ($p < 0.12$). Permanganate oxidizable carbon (POXC) at the 0- to 10-cm depth (panel B) by forest and treatment with post-analysis results on the right panel. Uppercase letters indicate significance ($p < 0.05$). Treatments are control (C, green), subsoiling (S, yellow), biochar amendment (B, gray), and biochar and subsoiling combination (B × S, purple).

potential to also improve drainage and aeration (Karim et al., 2020). In this study, similar trends were observed when comparing biochar-amended soil to the control. The reduction of bulk density is likely due to the addition of biochar as a low-density material. Biochar's high volume to mass ratio from abundant pore space would explain the 18% reduction in mean bulk density from control to biochar at the surface. This reduction is in line with other studies regarding reduction in bulk density using biochar of 1%–20% (Blanco-Canqui, 2017, Karim et al., 2020), with more compacted soils usually having larger responses (Karim et al., 2020). This increase in porous material has the potential to improve soil fertility and thus potential vegetation establishment through enhanced water movement, storage, and gas exchange. Similar results were observed when comparing the B × S treatment to the control, with a 19% decrease in mean bulk density at 0–10 cm. Both biochar amended treatments also included a significant increase in P, and trending increases in exchangeable cations when compared to the control.

Due to the abundance of above- and belowground obstacles, such as stumps or large diameter slash, incorporating biochar into the mineral soil can be costly and impractical for most logging operations in the US Midwest. For this reason, a broadcast application of biochar, without mixing into the soil, is a lower effort alternative. Over time, the biochar will be incorporated into the mineral soil through insect or animal burrowing and wetting–drying or freeze-thaw cycles. Random sampling combined with broadcasting biochar onto the soil surface likely played a significant role in the variability of soil measurement values. We observed that sheet flow from rainwater carried some of the biochar, causing it to be redistributed and concentrated in low spots within landings. The wide variation that we observed, while capturing overall trends, led to our inability to identify significant soil treatment effects. To counteract the effects of water and wind erosion, use of biochar in pellet form or of larger particle size could reduce the movement of the material as seen in Liao et al. (2022) and Roofchaee et al. (2024).

The effects of biochar treatments were mostly observed in the soil surface and decreased with depth, showing the potential limitations of top-dressed applications of this amendment during the first few years. Surface application of biochar likely played a role in this, as some studies estimate that biochar moves downward in the soil profile at rates ranging from 2 up to 30 mm year$^{-1}$ depending on soil characteristics, climate, and organisms present (Rumpel et al., 2015). Therefore, it would likely take more than a year for biochar to incorporate naturally to greater depths. Subsoiling appears to help incorporate biochar to greater depths more rapidly but the effect of the B × S treatment was not statistically significant at greater depths.

Subsoiling alone had little impact when compared to the control at the 0- to 10-cm depth. Obstacles that are common in landing soils such as stumps, large roots, and large diameter buried slash require wider tine spacing and force the operator to avoid them, resulting in noncontinuous subsoiling and sections of soil missed that contribute to an overall higher bulk density.

Because this study takes place over the course of only 1 year, changes in soil characteristics are potentially not entirely representative of the final outcomes of the treatments. Often, the surface application and the nature of forest sites may require decades of measurements to enable detection of greater soil changes (Page-Dumroese et al., 2025). As noted, surface-applied biochar will likely continue to move into the soil profile as water infiltration and vegetation establishment increase over time. These changes could lead to observable changes deeper in the soil over time.

## 5 | CONCLUSIONS

Surface applied biochar amendments at 22.4 Mg ha$^{-1}$ reduced compaction by roughly 18%–19% in the top 0–10 cm of soil, while also increasing other soil fertility metrics such as active C, plant available P, and exchangeable base cations. Subsoiling had little measured effect on soil characteristics when compared to the control and did not appear to incorporate biochar to a measurable degree. Lack of measurable subsoiling influence could partially be due to width and pathing of the subsoiler, as the operator moved to avoid underground obstacles. Our results indicate that biochar applications are beneficial for log landing soil rehabilitation. Additional research examining the effect of biochar particle size or pellets could be insightful for determining the influence of this factor on the displacement and effectiveness of biochar in landings, and subsequent effects on reestablishment of native vegetation and control of erosion or sedimentation. Longer term sampling on these sites could also be beneficial for understanding the temporal nature of biochar movement down compacted profiles in log landing soils.

## AUTHOR CONTRIBUTIONS

**William R. Rumpf**: Conceptualization; data curation; formal analysis; methodology; validation; visualization; writing—original draft; writing—review and editing. **Morgan P. Davis**: Conceptualization; data curation; formal analysis; funding acquisition; investigation; methodology; project administration; resources; software; supervision; validation; visualization; writing—original draft; writing—review and editing. **John M. Kabrick**: Conceptualization; formal analysis; methodology; resources; writing—review and editing. **John Stanovick**: Methodology; software. **Lauren S. Pile Knapp**: Conceptualization; supervision; writing—review and editing. **Deborah Page-Dumroese**: Conceptualization; methodology; writing—review and editing.

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## CONFLICT OF INTEREST STATEMENT

The authors confirm that there are no conflicts of interest.

## ORCID

*Morgan P. Davis* https://orcid.org/0000-0003-1310-2201
*Deborah Page-Dumroese* https://orcid.org/0000-0002-3095-7749

## REFERENCES

Andraski, B. J. (1991). Balloon and core sampling for determining bulk density of alluvial desert soil. *Soil Science Society of America Journal*, *55*(4), 1188–1190. https://doi.org/10.2136/sssaj1991.03615995005500040048x

Andrus, C. W., & Froehlich, H. A. (1983). *An evaluation of four implements used to till compacted forest soils in the Pacific Northwest* (Research Bulletin 45). Oregon State University.

Binkley, D., & Fisher, R. F. (2019). *Ecology and management of forest soils*. John Wiley & Sons.

Blanco-Canqui, H. (2017). Biochar and soil physical properties. *Soil Science Society of America Journal*, *81*(4), 687–711. https://doi.org/10.2136/sssaj2017.01.0017

Blanco-Canqui, H. (2021). Does biochar improve all soil ecosystem services?. *GCB Bioenergy*, *13*(2), 291–304.

Bray, R. H., & Kurtz, L. T. (1945). Determination of total, organic, and available forms of phosphorus in soils. *Soil Science*, *59*(1), 39–46.

Cambi, M., Certini, G., Neri, F., & Marchi, E. (2015). The impact of heavy traffic on forest soils: A review. *Forest Ecology and Management*, *338*, 124–138. https://doi.org/10.1016/j.foreco.2014.11.022

Coleman, R. A. (2012). Using response-ratios for tests of physiological hypotheses in field experiments. *Journal of Experimental Marine Biology and Ecology*, *428*, 1–4. https://doi.org/10.1016/j.jembe.2012.06.001

Dumroese, R. K., Page-Dumroese, D. S., & Pinto, J. R. (2020). Biochar potential to enhance forest resilience, seedling quality, and nursery efficiency. *Tree Planters' Notes*, *63*, 61–68.

Enyart, D., Aguilar, F., Kurtz, B., & Hernandez, R. (2009). *Missouri woody biomass harvesting best management practices manual*. Missouri Department of Conservation.

Franzluebbers, A. J. (2018). Short-term C mineralization (aka the flush of $CO_2$) as an indicator of soil biological health. *CABI Reviews*, *13*, 1–14. https://doi.org/10.1079/PAVSNNR201813017

Grether, C. H., & Creagh, K. (2018). *Michigan forestry best management practices for soil and water quality*. Michigan Department of Natural Resources.

Han, S. K., Han, H. S., Page-Dumroese, D. S., & Johnson, L. R. (2009). Soil compaction associated with cut-to-length and whole-tree harvesting of a coniferous forest. *Canadian Journal of Forest Research*, *39*(5), 976–989. https://doi.org/10.1139/X09-027

Hatchell, G. E., Ralston, C. W., & Foil, R. R. (1970). Soil disturbances in logging: Effects on soil characteristics and growth of loblolly pine in the Atlantic coastal plain. *Journal of Forestry*, *68*(12), 772–775.

Hedges, L. V., Gurevitch, J., & Curtis, P. S. (1999). The meta-analysis of response ratios in experimental ecology. *Ecology*, *80*(4), 1150–1156. https://doi.org/10.1890/0012-9658(1999)080[1150:TMAORR]2.0.CO;2

Homyak, P. M., Yanai, R. D., Burns, D. A., Briggs, R. D., & Germain, R. H. (2008). Nitrogen immobilization by wood-chip application: Protecting water quality in a northern hardwood forest. *Forest Ecology and Management*, *255*(7), 2589–2601. https://doi.org/10.1016/j.foreco.2008.01.018

Jackson, M. M., Turner, M. G., & Pearson, S. M. (2014). Logging legacies affect insect pollinator communities in southern Appalachian forests. *Southeastern Naturalist*, *13*(2), 317. https://doi.org/10.1656/058.013.0213

Karim, M. R., Halim, M. A., Gale, N. V., & Thomas, S. C. (2020). Biochar effects on soil physiochemical properties in degraded managed ecosystems in northeastern Bangladesh. *Soil Systems*, *4*(4), 69.

Keller, M., Varner, R., Dias, J. D., Silva, H., Crill, P., de Oliveira, Jr. R. C., & Asner, G. P. (2005). Soil–atmosphere exchange of nitrous oxide, nitric oxide, methane, and carbon dioxide in logged and undisturbed forest in the Tapajos National Forest, Brazil. *Earth Interactions*, *9*(23), 1–28.

Kozlowski, T. T. (1999). Soil compaction and growth of woody plants. *Scandinavian Journal of Forest Research*, *14*(6), 596–619. https://doi.org/10.1080/02827589908540825

Lee, M. R., McNeil, D. J., Mathis, C. L., Grozinger, C. M., & Larkin, J. L. (2021). Microhabitats created by log landings support abundant flowers and insect pollinators within regenerating mixed-oak stands in the central Appalachian Mountains. *Forest Ecology and Management*, *497*, 119472. https://doi.org/10.1016/j.foreco.2021.119472

Lehmann, J., & Joseph, S. (Eds.). (2015). *Biochar for environmental management: Science, technology and implementation*. Routledge.

Liao, W., Sifton, M. A., & Thomas, S. C. (2022). Biochar granulation reduces substrate erosion on green roofs. *Biochar*, *4*, 61.

Littell, R. C., Milliken, G. A., Stroup, W. W., Wolfinger, R. D., & Schabenberger, O. (2006). *SAS for mixed models* (2nd ed.). SAS Institute.

McNabb, D. H. (1994). Tillage of compacted haul roads and landings in the boreal forests of Alberta, Canada. *Forest Ecology and Management*, *66*(1), 179–194. https://doi.org/10.1016/0378-1127(94)90156-2

Miles, P. D., & Smith, W. B. (2009). *Specific gravity and other properties of wood and bark for 156 tree species found in North America* (Res. Note NRS-RN-38). USDA Forest Service, Northern Research Station. https://doi.org/10.2737/NRS-RN-38

Novak, J. M., Sigua, G. C., Watts, D. W., Cantrell, K. B., Shumaker, P. D., Szogi, A. A., Johnson, M. G., & Spokas, K. A. (2016). Biochars impact on water infiltration and water quality through a compacted subsoil layer. *Chemosphere*, *142*, 160–167. https://doi.org/10.1016/j.chemosphere.2015.06.038

Page-Dumroese, D. S., Jurgensen, M. F., Miller, C. A., Tirocke, J. M., Pierson, D. N., Mayn, C., Kimsey, M. J., & Anderson, H. C. (2025). Soil amendments alter tree growth and wood decay after forest thinning. *Soil Science Society of America Journal*, *89*(1), e20782.

Page-Dumroese, D. S., Jurgensen, M. F., Tiarks, A. E., Ponder, F., Jr., Sanchez, F. G., Fleming, R. L., Kranabetter, J. M., Powers, R. F., Stone, D. M., Elioff, J. D., & Scott, D. A. (2006). Soil physical property changes at the North American long-term soil productivity study sites: 1 and 5 years after compaction. *Canadian Journal of Forest Research*, *36*(3), 551–564. https://doi.org/10.1139/x05-273

Page-Dumroese, D. S., Tirocke, J. M., Anderson, N. M., Archuleta, J. G., McCollum, D. W., Morisette, J., Pierson, D. N., & Rodriguez-Franco, C. (2024). Continuous in-woods production of biochar using a trailer-mounted air curtain burner. *Journal of Visualized Experiments*, *206*, e66716.

Plotnikoff, M. R., Bulmer, C. E., & Schmidt, M. G. (2002). Soil properties and tree growth on rehabilitated forest landings in the interior cedar hemlock biogeoclimatic zone: British Columbia. *Forest Ecology and Management*, *170*(1), 199–215. https://doi.org/10.1016/S0378-1127(01)00776-9

Puettmann, M., Sahoo, K., Wilson, K., & Oneil, E. (2020). Life cycle assessment of biochar produced from forest residues using portable systems. *Journal of Cleaner Production*, *250*, 119564.

Rodriguez-Franco, C., & Page-Dumroese, D. S. (2021). Woody biochar potential for abandoned mine land restoration in the US: A review. *Biochar*, *3*(1), 7–22.

Rodriguez Franco, C., Page-Dumroese, D. S., Pierson, D., & Nicosia, T. (2024). Biochar utilization as a forestry climate-smart tool. *Sustainability*, *16*(5), 1714.

Roofchaee, A. S., Abrishamkesh, S., Fazeli, M., & Shabanpour, M. (2024). Optimizing biochar application: Effects of placement method, particle size, and application rate on soil physical properties and soil loss. *Journal of Soils and Sediments*, *24*(4), 1541–1555.

Rumpel, C., Leifeld, J., Santin, C., & Doerr, S. H. (2015). Movement of biochar in the environment. In J. Lehmann & S. Joseph (Eds.), *Biochar for environmental management* (pp. 283–300). Routledge.

Sidle, R. C. (1980). *Impacts of forest practices on surface erosion*. Oregon State University Extension Service.

Thomas, S. C., & Gale, N. (2015). Biochar and forest restoration: A review and meta-analysis of tree growth responses. *New Forests*, *46*(5), 931–946. https://doi.org/10.1007/s11056-015-9491-7

Thomas, S. C., Gorgolewski, A. S., & Vantellingen, J. (2022). Managed forests and methane: recent research and prospects for best management practices. In *Forest management for climate change mitigation: Recent innovations and research needs* (pp. 221–258). Springer.

Tucker, J. W., & Olson, D. P. (1994). Birds at log landings in the White Mountain National Forest, New Hampshire. *Maine Naturalist*, *2*(2), 91. https://doi.org/10.2307/3858252

USDA-NRCS. (2025). *Web soil survey*. http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm

Vantellingen, J., & Thomas, S. C. (2021). Log landings are methane emission hotspots in managed forests. *Canadian Journal of Forest Research*, *51*(12), 1916–1925.

Verheijen, F., Jeffery, S., Bastos, A. C., van der Velde, M., & Diafas, I. (2010). *Biochar application to soils: A critical scientific review of effects on soil properties, processes and functions*. European Commission, Publications Office. https://doi.org/10.2788/472

Weil, R. R., Islam, K. R., Stine, M. A., Gruver, J. B., & Samson-Liebig, S. E. (2003). Estimating active carbon for soil quality assessment: A simplified method for laboratory and field use. *American Journal of Alternative Agriculture*, *18*(1), 3–17. https://doi.org/10.1079/AJAA200228

Williamson, J. R., & Neilsen, W. A. (2000). The influence of forest site on rate and extent of soil compaction and profile disturbance of skid trails during ground-based harvesting. *Canadian Journal of Forest Research*, *30*(8), 1196–1205. https://doi.org/10.1139/x00-041

Wilson, K., Bekker, W., Archuleta, J., McAvoy, D., & Page-Dumroese, D. S. (2024). *Mobile biochar production by flame carbonization: Reducing wildfire risk and improving forest resilience* (Gen. Tech. Rep. RMRS-GTR-439). USDA Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-439

Zenner, E. K., Fauskee, J. T., Berger, A. L., & Puettmann, K. J. (2007). Impacts of skidding traffic intensity on soil disturbance, soil recovery, and aspen regeneration in north central Minnesota. *Northern Journal of Applied Forestry*, *24*(3), 177–183. https://doi.org/10.1093/njaf/24.3.177

## SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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