Biochar amendment as a tool for improving soil health and carbon sequestration in agro-ecosystems

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ABSTRACT

Conventional farming practices and land-use conversions drive carbon out of soil and into the atmosphere, where it contributes to climate change. Biochar, a soil amendment produced by pyrolyzing organic feedstocks under low-oxygen conditions, is a promising tool to restore soil carbon and draw down atmospheric carbon dioxide. Biochar has received considerable attention from scientists, growers, and environmentalists in the last 20 years, but there is still a gap between academic research and practical recommendations on biochar production and application that are relevant to small-scale growers. Here I present the results from two complementary studies that demonstrate the utility of local-scale biochar systems and provide some recommendations for those looking to work with biochar. The first study sought to determine the impact of biochar amendments on soil carbon and nutrient retention on three working farms across a variety of soil types, cropping systems, and climates in the United States. The effect of biochar amendment depended on initial soil characteristics and the properties of the biochar applied. Biochar amendments increased soil carbon in all three sites and increased soil nitrogen at two of the three. In this study pyrolysis conditions appeared to be as important as local soils and climate influences on the efficacy of biochar treatments. The second study was a life cycle assessment using SimaPro software to quantify the carbon balance and global warming potential of biochar produced from three local feedstocks (softwood, hardwood, and hay) applied to pasture soils in Southwest Virginia. Feedstock type, pyrolysis gas yield, and transportation distance significantly contributed to variation in the carbon balance of each agro-ecosystem. Biochar made from softwood lumber scraps performed best, with the highest net carbon storage and lowest global warming potential, followed by biochar made from hardwood scraps. Hay biochar performed worst, with positive carbon emissions (i.e., more carbon released than stored over its life cycle) in most scenarios tested, mainly because of its low biochar yield and the carbon emissions associated with agronomic production and transportation. Together these studies demonstrate the potential of local biochar systems to improve both soil health and carbon sequestration, and reinforce how important it is to know the characteristics of the soil and the

production history and properties of the biochar being applied in order to meet soil health and carbon sequestration goals.

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GENERAL AUDIENCE ABSTRACT

Conventional farming practices break down organic material in the soil, which decreases the capacity of soils to sustain crop growth and contributes to climate change as the soil releases carbon dioxide and other greenhouse gasses into the atmosphere. Biochar, or charcoal that is deliberately incorporated into soil, is gaining popularity among farmers, gardeners, and climate scientists for its ability to improve soil health and draw carbon out of the atmosphere to create stable long-term pools of carbon underground. Unfortunately, much of the research on biochar does not translate easily into recommendations for growers and land-managers to make and use biochar. Here I discuss the results from two studies examining the effect of biochar on soil health and carbon sequestration on local scales. In the first experiment I analyzed soil samples shared by farmers in New Mexico, Minnesota and Virginia who applied locally-sourced biochar to their soils. I found that the initial characteristics of the soil and of the biochar affected how the biochar application changed agriculturally-relevant soil properties. In general, biochar improved soil carbon and nitrogen levels, had mixed effects on soil pH depending on the biochar's pH, and had no effect on electrical conductivity (a measure of soil salinity). The second study was a life cycle assessment that quantified and compared greenhouse gas emissions of three different types of biochar, from feedstock harvest to biochar application to soil. I found that the type of feedstock used to make biochar, the amount of gas emitted during the conversion process, and the distance the feedstocks and biochar were transported all played a role in the overall carbon balance of the life cycle. The biochar made from softwood scraps performed best from a carbon storage perspective, followed by biochar made from hardwood. These two biochars tended to return more carbon to the soil than they emitted over their life cycle. The biochar made from hay performed worst, and emitted more carbon than it stored in most of the scenarios I tested. Together these studies show the potential of local biochar systems to improve both soil health and carbon sequestration and reinforce how important it is to be familiar with the soil and the production history and properties of the biochar being applied in order to meet soil health and carbon sequestration goals

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CHAPTER 1: Introduction

The balance of carbon storage in the atmosphere versus the biosphere is one of the most significant challenges we face as a species today. Nearly 200 years of fossil fuel combustion and land use change have increased the atmospheric carbon dioxide concentration by 50% above pre-industrial levels, resulting in warmer global temperatures and more intense weather events (Friedlingstein et al. 2022, NOAA 2021). Climate change poses a particular challenge for agriculture. Droughts, floods, pests, and disease are increasingly impacting the harvests and livelihoods of millions of people annually, especially in low- and middle-income countries (FAO 2021). At the same time, conventional agricultural practices like tilling and monoculture farming have driven the loss of enormous amounts of carbon from soil, leading to a global "carbon debt" (Sanderman et al. 2017, Goldstein et al. 2020, Zomer et al. 2017). This loss of soil carbon has severe consequences for soil health and crop yield and for global warming (Oldfield et al. 2021, Lal 2008).

Biochar, a soil amendment produced by pyrolyzing an organic feedstock under low-oxygen conditions, is a promising tool to take carbon out of the atmosphere and return it to soils. Biochar has received considerable attention from scientists, growers, and environmentalists in the last 20 years as a means for building long-lasting pools of soil carbon and improving other facets of soil health, like water-holding capacity, cation exchange, nutrient retention, and possibly even pathogen resistance (Lehmann et al. 2006). Biochar is also appealing because it can be made relatively cheaply with minimal equipment and from almost any organic feedstock (e.g., wood, hay, stover, slash, vinasse, silage, nut hulls, poultry waste, etc.), making it an accessible option for growers and land managers who want to incorporate it into their management plans at minimal cost. This accessibility also creates management challenges. With a wide array of pyrolysis techniques and feedstocks to choose from, it is important for potential biochar producers and consumers to have the information they need to create a biochar system that will meet their carbon sequestration and soil health goals.

The objectives of this research are to determine the impact of biochar amendment on soil carbon and nutrient retention in agricultural systems across a variety of soil types, cropping systems, and climates in the United States, and to quantify the "carbon cost" of biochar produced from different local feedstocks at one experimental site in southwestern Virginia using a life cycle assessment. The second chapter presents data from a field experiment comparing the effects of biochar on working farms in New Mexico, Minnesota, and Virginia. The third chapter comprises a life cycle assessment performed to compare the carbon emissions, global warming potential, and impacts on soil carbon storage potentials of biochar made from three different feedstocks (softwood, hardwood, and hay), and applied to a pasture system in southwest Virginia. Together these studies provide an assessment of the potential value of locally-sourced biochar for small- to mid-size farms, both in terms of soil health and carbon sequestration. Along the way are some recommendations based on the successes and shortcomings of the biochar systems explored here, which may prove useful to those interested in starting their own biochar journey.

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CHAPTER 2: Influence of biochar on soil health across a range of soils and agro-ecosystems

Abstract

Application of black carbon to agricultural soils has been used to boost soil fertility for millennia. In the past few decades farmers and land managers have begun to use biochar, a soil amendment produced by pyrolyzing organic feedstocks under low-oxygen conditions, to improve soil health and create stable, long-lasting pools of carbon in soil. While there is a broad consensus that biochar can improve a range of important physical and biological soil properties, there is still a gap between academic research and recommendations on biochar production and application for growers. We collaborated with growers in New Mexico, Minnesota, and Virginia to establish biochar field experiments on working farms and examine the impact of biochar amendment on carbon sequestration and several important soil characteristics (soil organic carbon, soil nitrogen, pH, and electrical conductivity) in agricultural systems across a variety of soil types, cropping systems, and climates in the United States. At the highest biochar application level (2.5 kg/m^2), I saw increases in soil carbon of 1.67, 1.5, and 1.35x the control in Virginia, Minnesota, and New Mexico, respectively. Soil nitrogen increased significantly in Virginia, suggesting an interactive effect between the biochar and blood meal fertilizer that was co-applied. The overall pH effect was driven by the Minnesota site, where the acidic, under-pyrolyzed biochar significantly decreased pH, and there was no effect on electrical conductivity at any site. Taken together, these results provide some suggestions and warnings for growers planning to introduce biochar to an agro-ecosystem. A higher rate of biochar application (at least up to 2.5 kg/m^2) provides better results, co-application of biochar with an organic fertilizer may help boost soil nitrogen, and incomplete pyrolysis can create acidic biochar that drives down soil pH. Above all, it is crucial to understand the characteristics of both soil and biochar before amendment to achieve the desired result and avoid unintended consequences.

Introduction

The history of black carbon in agricultural soils stretches back millennia. In the Amazon rainforest, patches of soil called *terra preta do indio* are darker, higher in carbon, and more fertile than the surrounding highly weathered tropical soils (Glaser et al. 2001). These rich soils formed over time as indigenous communities incorporated charcoal and organic waste materials into the soil. Farmers around the world have also long practiced swidden agriculture, in which land is burned, cultivated, and left fallow on a cycle of years to decades (Nye and Greenland 1960). In both these methods, the black carbon, or charcoal, produced from burning plant material contributes to higher and more persistent soil carbon content, which in turn improves soil fertility (Schmidt et al. 2021).

More recently, scientists and land managers have considered biochar applications as a method for realizing the benefits of black carbon. Biochar is a soil amendment produced by pyrolyzing organic feedstocks under low-oxygen conditions (Lehmann et al. 2006). In the past, farmers have primarily been interested in the soil fertility benefits of black carbon; today, interest in biochar has expanded to include its role in carbon management as a tool to offset anthropogenic CO_2 emissions and combat climate change. Agricultural systems are a particularly fertile field for carbon sequestration strategies, as tillage and erosion have caused a global soil "carbon deficit" (Sanderman et al. 2017, Goldstein et al. 2020, Six et al. 1998, Zomer et al. 2017). Biochar is therefore a promising tool for drawing down atmospheric carbon, restoring soil organic carbon, and improving soil fertility in agricultural systems (Lehmann et al. 2006, Woolf et al. 2010, Downie et al. 2012).

There is broad consensus in the scientific literature that biochar amendments, on average, have positive effects on agriculturally important soil properties. Biochar addition has been shown to increase soil carbon content (Weng et al. 2017, Ji et al. 2016, Yoo et al. 2017, Li et al. 2019); water retention (Blanco-Canqui 2017, Kavitha et al. 2018, Bruun et al. 2014, Kumar et al. 2020); soil aggregation and physical structure (Blanco-Canqui 2017); soil nutrients, especially nitrogen and phosphorus (Kavitha et al. 2018, Biederman and Harpole 2013, Li et al. 2019); and plant productivity and crop yield, likely as a result of all these physical and chemical changes (Atkinson et al. 2010, Jeffery et al. 2017, Kavitha et al. 2018, Biederman and Harpole 2013, Ji et al. 2016, Ibrahim et al. 2019, Bruun et al. 2014). While many studies suggest that biochar may be useful for mitigating soil degradation and soil organic carbon loss (Allohverdi et al. 2021, Li et al. 2019, Weng et al. 2017, Bruun et al, 2014, Blanco-Canqui 2017), biochar research is still developing primarily within academic settings and many questions remain. In particular, there is a gap between the research being conducted into biochar's effect on soil and specific guidelines on pyrolysis, preparation, and rates of application geared towards the growers and managers interested in using it. There are not enough studies done under realistic agricultural conditions across a broad range of crops and soil types, limiting the utility of scientific literature for the farmers and land managers who could benefit from it most.

The objectives of this research are to determine the impact of biochar amendment on carbon sequestration and several important soil characteristics in agricultural systems across a variety of soil types, cropping systems, and climates in the United States. Ideally, farmers and land managers can use these results to make informed decisions around biochar and anticipate the costs and benefits of applying biochar to their particular system.

Methods

Site Descriptions and Biochar Application

Sites were chosen to represent a range of soil types, climates, and cropping systems. All sites are located on small- to medium-size family farms (5 - 600 acres) in New Mexico, Minnesota, and Virginia. Figure 1 shows the location, elevation, mean annual temperature, mean annual precipitation, and crop grown at each site. Growers established four 20m^2 experimental plots replicated four times in a randomized split-plot design. The plots varied in shape depending on the size and planting scheme of each site, but all plots at a given location were consistent in the crop planted and the management system in place. Plots were each assigned an amendment: high biochar (2.5 kg/m^2 or 25 tonnes/hectare) with 0.11 kg/m^2 blood meal co-application, low biochar (1 kg/m^2 or 10 tonnes/hectare) with 0.11 kg/m^2 blood meal co-application, blood meal alone (0.11 kg/m^2) as a positive control, and no biochar or blood meal as a negative control. Blood meal, a nitrogen-rich fertilizer made from dried animal blood, was co-applied with the biochar to minimize any stimulation of nitrogen immobilization from adding biochar alone. The decision to co-apply blood meal was made with the input of our grower collaborators, who wanted to preempt any nutrient deficiencies during the first year of the experiment. Treatments were applied in the spring or summer of 2019 and all plots, including the controls, were tilled, disced, or turned, depending on the growers' equipment and crops, to incorporate the treatments into the soil. Growers then proceeded with their usual planting and maintenance scheme on the experimental plots.

Biochar Production

The biochar used at each site was produced locally using waste materials from nearby agricultural processes as feedstock. Table 1 provides information on the production and properties of biochar used at each site.

Soil Sampling

Once before and at least once after biochar application, growers collected 5 samples from each treatment plot using a soil corer or trowel to excavate the top 10cm of soil. The samples were taken from points spaced out across the plots to capture spatial variability. Aboveground plant material (for example, grasses or stalks) was removed, but belowground roots were retained in the samples. The 5 samples were mixed together in a bucket and transferred to paper bags for drying. The mixing bucket was rinsed and dried between plots if needed to avoid transfer of biochar or blood meal. The composite samples were air-dried for approximately 48 hours in a cool, dry place. Once dry, a representative subsample was transferred to a gallon-sized ziplock freezer bag and shipped to Virginia Tech for analysis.

Soil Analysis

Once in the lab, the soil samples were passed through a 4mm sieve to remove rocks, pieces of plant material, and large pieces of biochar, which were retained. The fine earth fraction of soils was then analyzed for pH, electrical conductivity (EC), total organic carbon content, and total nitrogen content.

Soil pH is an important physicochemical property that can affect a range of other properties, including nutrient retention and crop productivity. To measure pH, 10g of air dried soil was mixed with 20mL of deionized water in an Erlenmeyer flask or small Nalgene bottle, shaken into a slurry, and then left to sit for 10 minutes. A Hach sensION + pH meter (Hach Company, Loveland, CO, USA) was calibrated with standard solutions and then placed into the soil slurry until the reading stabilized. The probe was rinsed with deionized water between samples.

Electrical conductivity (EC) is a measure of the ionic activity, e.g., salinity, in a soil sample. Water soluble ion content can impact plant, fungal, and microbial growth in soils, as well as water and nutrient availability (USDA NRCS 2014). To measure EC, an additional 30mL of deionized water was added to the slurries used for pH analysis to create a 1:5 soil to water dilution. This mixture was shaken for 5-10 seconds and then left to sit for 10 minutes. A YSI 3100 Conductivity Instrument (YSI Company, Yellow Springs, OH, USA) was calibrated using a standard solution of 0.01M KCl. The probe was placed into each flask or bottle while the slurry was gently swirled, and the value was recorded when the instrument's reading stabilized. The probe was rinsed with deionized water between samples.

Total soil organic carbon is a key indicator of soil health, especially in agricultural systems (FAO 2015). Soil carbon improves soil structure, aeration, water movement and retention, biodiversity, and crop yield (Trivedi et al. 2018). Soil is also an enormous carbon pool on a global scale, and restoring this pool to offset carbon emissions into the atmosphere is one of the main reasons for applying biochar to soils. To measure total soil organic carbon, air dried samples were ground into a fine powder using a SPEX SamplePrep Mixer/Mill and run on an Elementar Vario MAX Cube CNS analyzer. The samples from New Mexico, where the arid climate causes inorganic carbon to accumulate in soil, were acid-fumigated before carbon content analysis to drive off carbonates (Suarez 2006, Dhillon et al. 2015, Harris et al. 2001). The soils from Minnesota and Virginia were not acidified.

Total soil nitrogen was also measured when the samples were run on the elemental analyzer. Nitrogen exists in many forms, both organic and inorganic, in soil, and is one of the most important elements for plant growth. While this analysis does not differentiate between forms of organic and inorganic nitrogen, total nitrogen content and the C:N ratios of a soil are good predictors of potential nitrogen availability in agricultural soils. In most soils, nitrogen exists primarily in organic forms, which microbes readily convert into plant-available inorganic forms (Robertson and VanderWulp 2019, Cornell SIPS 2020). This analysis was sufficient to assess how biochar and blood meal influenced the nitrogen content of soils.

Statistical Analysis

For each soil property of interest (i.e., pH, EC, total organic carbon, and total nitrogen) a two-way ANOVA (analysis of variance) test was performed. The independent variables were location (New Mexico, Minnesota, or Virginia) and treatment (high biochar + blood meal, low biochar + blood meal, blood meal alone, or no amendment). The results of the ANOVA indicated whether the treatments affected a given soil property, and whether this effect differed between locations. In all analyses, block number was a random effect to account for spatial variation within the sampling sites, and in the post-treatment analyses season was a random effect to account for variation between timepoints that was not due to the treatments. Post hoc Tukey's HSD tests were done for each location and time point to identify significant differences between individual treatments.

Results

Soil Total Carbon

Soil organic carbon is the property where I expected to see the largest change, as biochar is a carbon-rich amendment. As expected, I did find an overall increase in soil carbon, with notable differences among locations. The baseline carbon contents of soils at each site before biochar amendments were applied were significantly different (location effect, $p = 3.139e-05$), but the differences among plots at each site were not (treatment effect, $p = 0.1679$; treatment:location effect, $p = 0.9420$; see Table 2 and Figure 2a). The New Mexico soils averaged 1.75 ± 0.11% carbon, the Minnesota soils averaged 2.38 ± 0.03% carbon, and the Virginia soils averaged 2.81 ± 0.07% carbon by weight (see Table 2 and Figure 2*a*).

After biochar was applied, the carbon content of both the high and low biochar treatments increased in all locations (treatment effect, $p = 1.875e-09$), but the sites differed in the effect size of the treatment (treatment:location effect, $p = 0.006419$; see Figure 2b). Virginia exhibited the largest increase in soil carbon, with approximately 1.67x higher soil carbon percentage in the high biochar treatment compared to the unamended control and 1.74x higher soil carbon in the high biochar soils compared to the blood meal-only soils. The control plots averaged 3.27 ± 0.19% carbon, the high biochar plots averaged 5.46 ± 0.98% carbon, the low biochar plots averaged 3.87 ± 0.34% carbon, and the blood meal plots averaged 3.14 ± 0.32% carbon. In Minnesota, the high biochar soils had about 1.5x the average soil carbon percentage of the control soils and 1.4x the average soil carbon percentage of the blood meal-only soils. The control plots averaged 2.53 ± 0.07% carbon, the high biochar plots averaged 3.8 ± 0.35% carbon, the low biochar plots averaged 3.27 ± 0.21% carbon, and the blood meal plots averaged 2.72 ±

0.07% carbon. New Mexico had the smallest increase, with around 1.35x the soil carbon percentage in the high biochar soils compared to both the unamended control and the blood meal-only soils. In New Mexico, the control plots averaged 1.47 ± 0.20% carbon, the high biochar plots averaged 1.98 ± 0.29% carbon, the low biochar plots averaged 1.86 ± 0.17% carbon, and the blood meal plots averaged 1.45 ± 0.19% carbon.

Soil Total Nitrogen

As with the other soil properties, the sites had significantly different baseline total soil nitrogen contents (location effect, p = 1.491e-05); however, there were also underlying differences among plots that contributed to a significant "treatment effect" even before amendments were applied; i.e., there was significant within-site spatial variation in soil nitrogen content (treatment effect, p = 0.02558; see Table 2 and Figure 3a). This appeared to be driven by the low initial nitrogen contents of the plots that would later be amended with blood meal in the Minnesota and Virginia sites compared with the high and low biochar plots at those sites, though the pairwise comparisons between those plots were not significant (p > 0.10 for both). In Minnesota, the control plots averaged 0.23 ± 0.02% nitrogen by weight, the high biochar plots averaged 0.25 ± 0.02% nitrogen, the low biochar plots averaged 0.23 ± 0.01% nitrogen, and the blood meal plots averaged 0.21 ± 0.02% nitrogen. In Virginia, the control plots averaged 0.22 ± 0.01% nitrogen, the high biochar plots averaged 0.24 ± 0.02% nitrogen, the low biochar plots averaged 0.24 ± 0.01% nitrogen, and the blood meal plots averaged 0.21 ± 0.01% nitrogen. In New Mexico, the control plots averaged 0.16 ± 0.02% nitrogen, the high biochar plots averaged 0.15 ± 0.02% nitrogen, the low biochar plots averaged 0.17 ± 0.01% nitrogen, and the blood meal plots averaged 0.15 ± 0.02% nitrogen. All post hoc pairwise comparisons of nitrogen content between plots at each location were not significant, yielding p > 0.10.

There was a marginal effect of biochar on total soil nitrogen after amendment (treatment effect, p = 0.1215, see Figure 3b) that was driven by the Virginia site, which had a significant increase in total soil nitrogen in the high biochar plots compared to both the unamended (negative control) and blood meal (positive control) plots. In Virginia, the control plots averaged 0.28 ± 0.02% nitrogen, the high biochar plots averaged 0.34 ± 0.05% nitrogen, the low biochar plots averaged 0.29 ± 0.02% nitrogen, and the blood meal plots averaged 0.26 ± 0.004% nitrogen. The post hoc pairwise comparisons between the high biochar treatment and the control, blood meal, and low plots had p values of 0.047, 0.03, and 0.05, respectively. Each of the other pairwise comparisons were insignificant (i.e., $p > 0.94$). In Minnesota the control plots averaged 0.22 ± 0.01% nitrogen, the high biochar plots averaged 0.24 ± 0.01% nitrogen, the low biochar plots averaged 0.24 ± 0.01% nitrogen, and the blood meal plots averaged 0.23 ± 0.01% nitrogen, and post hoc pairwise comparisons between treatments all had $p > 0.50$. In New Mexico, the control plots averaged 0.14 ± 0.02% nitrogen, the high biochar plots averaged 0.14 ± 0.01% nitrogen, the low biochar plots averaged 0.15 ± 0.01% nitrogen, and the blood meal plots averaged 0.13 ± 0.02% nitrogen. Post hoc pairwise comparisons between treatments all had $p > 0.77$.

Soil pH

Before treatments were applied, the average pH of the three locations (New Mexico, Minnesota, and Virginia) were significantly different from one other (location effect, p = 7.837e-06; see Table 2 and Figure 4a), reflecting different baseline soil properties due to local geology and climate. In the semi-arid New Mexico soils, pH values ranged from 6.63 - 8.41 with an average of 7.86. In Minnesota they were 7.18 - 7.69 with an average of 7.41. Virginia had the lowest baseline soil pH, with values from 5.62 - 6.61 and an average of 6.06 (see Table 2 and Figure 2*a*). Within each location, there was no significant pre-treatment difference in pH among the plots that would later be amended (treatment effect, $p = 0.807$; treatment:location effect, $p = 0.6434$).

After treatments were applied, there was a marginal effect on pH, though the effect size varied among locations (treatment effect, $p = 0.054$; treatment:location effect, $p = 0.115$; see Figure 4b). This trend appears to be driven by the pH changes at the Minnesota site, where the added biochar was not fully pyrolyzed and very acidic (see Table 1), and drove the pH of the high and low biochar treatment plots down well below the unamended control. In Minnesota the control plots averaged 7.63 ± 0.07, the low biochar treatment plots averaged 7.31 ± 0.06, the high biochar treatment plots averaged 7.33 ± 0.07, and the blood meal plots averaged 7.32 ± 0.06. Post hoc pairwise comparisons between the control plots and each other treatment at the Minnesota site were $0.08 > p > 0.06$. In Virginia, where the biochar used was alkaline (Table 1), there was a marginal increase in pH in the high biochar plot compared to the unamended control. The pH in the control plots averaged 5.56 ± 0.07, the low biochar plots averaged 5.52 ± 0.11, the high biochar plots averaged 5.81 ± 0.13, and the blood meal plots averaged a pH of 5.51 ± 0.13 (Figure 4a&b). In New Mexico, where the biochar applied was alkaline as well (Table 1), there was no effect of any treatment on soil pH; the control plots averaged 7.83 ± 0.25, the low biochar plots averaged 7.87 ± 0.30, the high biochar plots averaged 7.91 ± 0.22, and the blood meal plots averaged 7.68 ± 0.30. A post hoc pairwise comparison between the high biochar and blood meal plots had $p = 0.19$; all other pairwise comparisons had $p > 0.33$.

Soil Electrical Conductivity

As with soil pH, there was a statistically significant pre-treatment difference in electrical conductivity among locations (location effect, $p = .004$), as well as variation among plots at each location before treatments were applied (Treatment effect, $p = 0.088$; see Table 2 and Figure 5a). These differences appeared to be driven primarily by the high baseline electrical conductivity of soils from the New Mexico "low biochar" plots. In the New Mexico pre-treatment soils, the "control" plots averaged 157.8 ± 29.6µS, the "high biochar" plots averaged 127.2 ± 28.8µS, the "low biochar" plots averaged 198.3 ± 38.1µS, and the "blood meal" plots averaged 140.8 ± 28.4µS. Pairwise comparisons of "high" and "low" biochar plots vs. control plots yielded p values of 0.03 and 0.09, respectively, indicating a significant pre-treatment block effect. At the Minnesota site, the pre-treatment electrical conductivity values were as follows: the "control" plots averaged 153.1 ± 5.8µS, the "high biochar" plots averaged 142.3 ± 3.1µS, the "low biochar" plots averaged 142.5 ± 18.4µS, and the "blood meal" plots averaged 161.8 ± 11.3µS. In Virginia, the "control" plots averaged 53.0 ± 1.8µS, the "high biochar" plots averaged 53.0 ± 2.4µS, the "low biochar" plots averaged 79.0 ± 16.0µS, and the "blood meal" plots averaged 62.5 ± 13.7µS. Post hoc pairwise comparisons of treatments in Virginia and Minnesota were all non-significant ($p > 0.82$).

Post-treatment, there was no significant effect of any biochar or blood meal treatment on soil electrical conductivity. There continued to be a difference in electrical conductivity among locations (location effect, $p = 3.248e-05$), but the statistically significant differences that were previously seen within treatments at each location were no longer present (treatment effect, $p = 0.5936$; treatment:location effect, $p = 0.6833$; see Figure 5b). In New Mexico, the control plots averaged 127.6 ± 30.0µS, the high plots averaged 115.5 ± 20.3µS, the low biochar plots averaged 167.1 ± 48.4µS, and the blood meal plots averaged 125.6 ± 27.8µS. In Minnesota, the control plots averaged 133.2 ± 12.3μS, the high plots averaged 129.1 ± 12.0μS, the low biochar plots averaged 128.4 ± 12.7μS, and the blood meal plots averaged 123.7 ± 12.8μS. In Virginia, the control plots averaged 57.9 ± 5.8μS, the high plots averaged 74.9 ± 22.5μS, the low biochar plots averaged 51.5 ± 2.8μS, and the blood meal plots averaged 56.6 ± 6.8μS.

Discussion

I found that the effects of biochar on the soil properties I analyzed were context-dependent, influenced by the initial characteristics of the soil and the quality and composition of the biochar applied to it (see Table 3 for a summary of results). Indeed, identifying a variety of responses among sites was the goal of the study. Neither soil nor biochar are homogenous within or among locations; they are substances with properties that reflect the materials and processes that produce them. I chose to study a variety of soils on working farms in New Mexico, Minnesota, and Virginia, amended with a variety of locally-produced biochar, in order to assess the range of responses growers may expect from biochar applications on universally recognized and easy-to-measure properties of soil health.

Soil Organic Carbon

Increases in soil organic carbon, potentially the most important factor I looked at in terms of soil health and carbon sequestration, were the most consistent result across all of my sites. All locations exhibited increases in carbon at both the high (2.5 kg/m^2) and low (1 kg/m^2) rates of biochar addition, though only the high biochar plots had statistically significant increases above the unamended controls. Soils in the Virginia site had the largest increase, with 1.67x more soil carbon in the high biochar plots compared to the controls. The high plots in Minnesota had a carbon content 1.5x higher than the control plots, while the New Mexico soils had the smallest increase in carbon, at 1.35x higher in the high plots than the control plots. To put this in perspective, those increases equate to approximately 7600 kg/ha (3.4 US tons/acre) of carbon returned to the soil in Virginia, 5300 kg/ha (2.4 US tons/acre) of carbon in Minnesota, and 2800 kg/ha (1.25 US tons/acre) of carbon in New Mexico. At the time of this report, one ton of offset carbon is valued at just over $30 USD on the California Carbon Credit Market (CarbonCredits.com). On the farms studied here that means anywhere between $37.50 - $102 per acre in potential carbon credits. For reference, these numbers fall within the range of net profits observed and anticipated per acre of corn and soy planted in high-quality farmland in central Illinois from 2019-2021 (Schnitkey et al. 2021). Depending on the crop planted and current market prices, carbon credits from biochar amendment could potentially double a grower's profit per acre of farmland. The biggest financial benefit would come from biochar that is produced locally using agricultural waste materials that would otherwise be discarded. In addition to the value of the soil carbon itself, biochar amendment has also been shown to increase crop yields in some circumstances, producing additional revenue. It is important to know, however, that these benefits are most commonly seen in soils with lower initial fertility, such as tropical or degraded soils, and therefore may not be seen in soils that are already healthy and fertile (Keske et al. 2020, Jeffery et al. 2017).

My results were promising from the perspective of carbon sequestration potential as well. Each site saw an increase in soil organic carbon after biochar application, which represents a largely stable pool of carbon that would have otherwise entered the atmosphere through decomposition or combustion of waste materials. Woolf et al. (2010) estimated that over a century, biochar application to soils could reduce net greenhouse gas emissions globally by 12% without compromising food security, habitat, or soil health. This estimate relied on biochar amendments at a maximum rate of 50 tonnes/hectare (double the highest rate applied in this study) to agricultural soils. Biochar is certainly not a silver bullet for solving climate change, but sequestering carbon in agricultural soils could be one method in a multi-pronged climate change mitigation strategy.

Total Soil Nitrogen

Biochar amendment boosted total soil nitrogen in the high biochar Virginia plots, and showed no effect in New Mexico or Minnesota. The nitrogen content of the biochar itself did not appear to drive this effect, as the biochar used in New Mexico had the highest nitrogen content (0.89% by mass), followed by Virginia (0.66%) and Minnesota (0.47%) (see Table 1). Pre-amendment soil nitrogen content was also not a consistent predictor of post-amendment results, as the soils in Minnesota and Virginia showed different responses to biochar amendment despite their similar initial nitrogen content. It is important to remember here that nitrogen-rich blood meal was co-applied with biochar at all sites. Interestingly, the blood meal-only treatments had total soil nitrogen levels equal to or below those of the biochar and unamended control plots at all sites, suggesting that the blood meal nitrogen was rapidly consumed by plants or microbes. The only situation where blood meal nitrogen appeared to be retained was when it was co-applied with biochar in Virginia, where the increase in soil nitrogen in the high biochar plots was larger than would be expected by the addition of the nitrogen within the biochar itself (see Table 1). The increased soil nitrogen in the high biochar plots compared to the blood meal-only plots in Virginia suggest that the biochar may have helped to immobilize or otherwise slow the loss of nitrogen from the soil, which is a potential benefit that has been noted in other studies (Biederman and Harpole 2013, Hossain et al. 2020), consistent with the influence of high C:N ratio organic matter in nitrogen retention (Barrett and Burke 2000 & 2002, Burke et al. 2012). This mechanism could help explain the lack of effect in Minnesota, where the under-pyrolyzed, woody biochar lacked the porous structure and charged surface that causes biochar to hold onto nutrients in the soil (Weber and Quicker 2018).

Soil pH and Electrical Conductivity

The pH results were an interesting case study of biochar's context-dependent influence and potential soil improvements and also a valuable cautionary tale about under-pyrolyzed biochar. The soils considered in this study varied widely in pH, from well below 6 in the weathered Virginia pasture to above 8 in the arid New Mexico soils. The pH of the biochar itself varied widely as well: the hardwood biochar used in Virginia had a pH of 8.8, the pecan shell biochar used in New Mexico had a pH of 8.52, and the woodchip biochar used in Minnesota had a pH of 3.48 (Table 1). The overall treatment effect was driven by the very acidic biochar applied in Minnesota, which had the opposite of the liming effect typically associated with biochar amendment. In New Mexico the biochar had little to no effect, likely due to the negligible difference in pH between the alkaline pre-treatment soils and the alkaline biochar that was applied to them. Virginia appeared to be the best candidate for biochar management of soil pH: it had the lowest baseline soil pH and the highest biochar pH of the three locations. Overall, the pH results make it clear that it is important to be aware of both initial soil pH, influenced by previous liming or local geology, and the pH of biochar before deciding whether to amend soil with biochar in a particular location.

Biochar amendment had no apparent effect on electrical conductivity in the sites tested. Biochar has a physically and chemically complex structure, with both positively and negatively charged functional groups on the surface that can react with soluble ions in the soil matrix (Weber and Quicker 2018, Atkinson et al. 2010). Depending on the initial soil conditions and the characteristics of the biochar itself, biochar can be a sink or source of solutes (Rehrah et al. 2014). Field studies where both initial electrical conductivity and feedstocks were similar to the sites and biochars used here have shown this, with biochar amendment causing anywhere from a ~200% increase to a ~100% decrease in soil electrical conductivity (Vijay et al. 2021). Considering that the soils included in this study were not being negatively affected by salinity and the biochar did not appear to drive electrical conductivity to levels that would hamper plant growth, growers working in similar soils and with similar biochars likely do not need to worry about spikes in conductivity after biochar application.

Biochar Application Recommendations

Specific guidelines for biochar application remain difficult to find due to the wide range of initial soil and biochar characteristics and the lack of long-term field studies to draw from (Vijay et al. 2021). I offer the following recommendations based on the trends I observed across the three sites studied here:

1. First and foremost, it is crucial that growers and land managers know their soil, know their biochar, and know the outcome they are looking to achieve with biochar amendment (e.g., increased soil carbon, increased pH, or increased crop yield).
2. I found that the higher biochar application rate (2.5 kg/m^2 or 25 tonnes/hectare) produced the best results, specifically for soil carbon and soil nitrogen in Minnesota and Virginia. This application rate is near the midpoint of the range seen in many biochar studies (Schmidt et al. 2021, Major 2010). If a similar rate is economically and logistically feasible, I recommend it as a starting point that can be modified over time to achieve the desired outcome.
3. Given that co-applying biochar and an organic fertilizer had neutral-to-positive effects on soil nitrogen, and there was some evidence in Virginia that co-application actually improved retention of blood meal-derived nitrogen, I recommend that growers interested in incorporating biochar into their cropping systems do so in conjunction with their usual fertilizing scheme, or co-apply biochar with a fertilizer or compost. This method is

widely practiced and recommended in the scientific and agronomic literature (Kang et al. 2021, Schmidt et al. 2021)

4. If a liming effect is desired with biochar application, it is important to fully pyrolyze the feedstocks. The results from Minnesota, where the acidic biochar drove down soil pH, are a cautionary tale for growers looking to maintain or increase soil pH. Additionally, a liming effect can only be expected if the biochar applied has a pH well above that of the pre-treatment soil, meaning well-pyrolyzed biochar is unlikely to have a noticeable effect on pH in alkaline soils.

Conclusion

My goal for this study was to examine the effect of realistic application rates of locally-sourced biochar on working farm soils in New Mexico, Minnesota, and Virginia, which represented a range of soil types and cropping systems. I specifically looked at four properties that are useful indicators of soil health in agricultural systems: soil organic carbon, total soil nitrogen, soil pH, and soil electrical conductivity. I found that the effect of biochar amendment depended on initial soil characteristics and the properties of the biochar applied. The Virginia site saw the best outcomes, with increased soil carbon and nitrogen and marginally higher soil pH with a biochar application rate of 2.5 kg/m^2. Minnesota also saw an increase in soil carbon, but the under-pyrolyzed biochar applied there was acidic, and drove down soil pH. New Mexico, which had the lowest baseline soil carbon and nitrogen levels and highest initial pH and electrical conductivity values, showed little response to biochar application beyond a slight increase in soil carbon. Taken together, these results emphasize the importance of understanding baseline soil and biochar characteristics, as well as the desired outcome, before biochar application to agricultural soils. Biochar generally had neutral-to-positive effects across the three sites I studied, and is a promising tool for growers and land managers to improve soil properties and sequester carbon.

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Tables

Table 2.1. Information on pyrolysis and biochar physical and chemical characteristics. Feedstock and pyrolysis details were reported by producers, physical and chemical characteristics come from testing performed by Control Laboratories in Watsonville, CA. Note: The biochar used in Minnesota was a mixture of lightly and more heavily torrefied chars, combined to reach an average fixed carbon content of ~30%.

Location	Feedstock	Pyrolysis Method	Max. Temp (°C)	Time at Temp	Bulk Density (lb/cu ft)	Organic C (% of total dry mass)	Total N (% of total dry mass)	pH	Electrical Conductivity (uS)	Surface Area (m^2/g dry)	Particle Size Distribution (%)					
											< 0.5 mm	0.5-1 mm	1-2 mm	2-4 mm	4-8 mm	8-16 mm
New Mexico	Pecan hulls	Continuous TLUD	2200	>1 min	21.5	84.5	0.89	8.52	0.144	192	8.9	10.2	23	29.6	27.6	0.7
Minnesota	Red oak + pine chips	Torrefaction	250-275	10-20 min	16.8	52.5	0.47	3.48	0.075	140	0.2	0	0.3	7.3	49.6	42.5
Virginia	Hard-wood	Charcoal retort	500	2 hr	18.4	84.5	0.66	8.8	0.225	188	24.8	12.6	15	25.4	18.4	3.8

Table 2.2 Pretreatment mean values (± standard error) for pH, EC, total soil carbon, and total soil nitrogen by location.

Location	**Pre-treatment Mean pH**	**Pre-treatment Mean EC (microSiemens cm^{-1})**	**Pre-treatment Mean Total Carbon (%)**	**Pre-treatment Mean Total N (%)**
New Mexico	7.86 ± 0.17	156.01 ± 15.69	1.75 ± 0.11	0.16 ± 0.008
Minnesota	7.41 ± 0.04	149.93 ± 5.40	2.38 ± 0.03	0.23 ± 0.006
Virginia	6.06 ± 0.09	61.87 ± 5.55	2.81 ± 0.07	0.23 ± 0.007

Table 2.3 Schematic representation of biochar treatment effects. Arrow direction and color indicates an increase (green, up) or decrease (red, down) in the measured soil characteristic between the control and/or blood meal only and the high biochar treatment. Solid arrows indicate a statistically significant change, while open arrows indicate a non-statistically significant trend. The "~" symbol indicates no change.

Location	Total Soil Carbon	Total Soil Nitrogen	pH	Electrical Conductivity
New Mexico	⇧	~	⇧	~
Minnesota	⬆	~	⬇	~
Virginia	⬆	⬆	⇧	~

Figures

Figure 2.1. Map of experimental plot locations. Field sites were located on working farms in southwestern Virginia, southwestern Minnesota, and southwestern New Mexico. Elevation, mean annual temperature, and mean annual precipitation were collected from the Oregon State University PRISM Climate Group's explorer tool: https://prism.oregonstate.edu/explorer/. Crop types were reported by the growers in 2019.

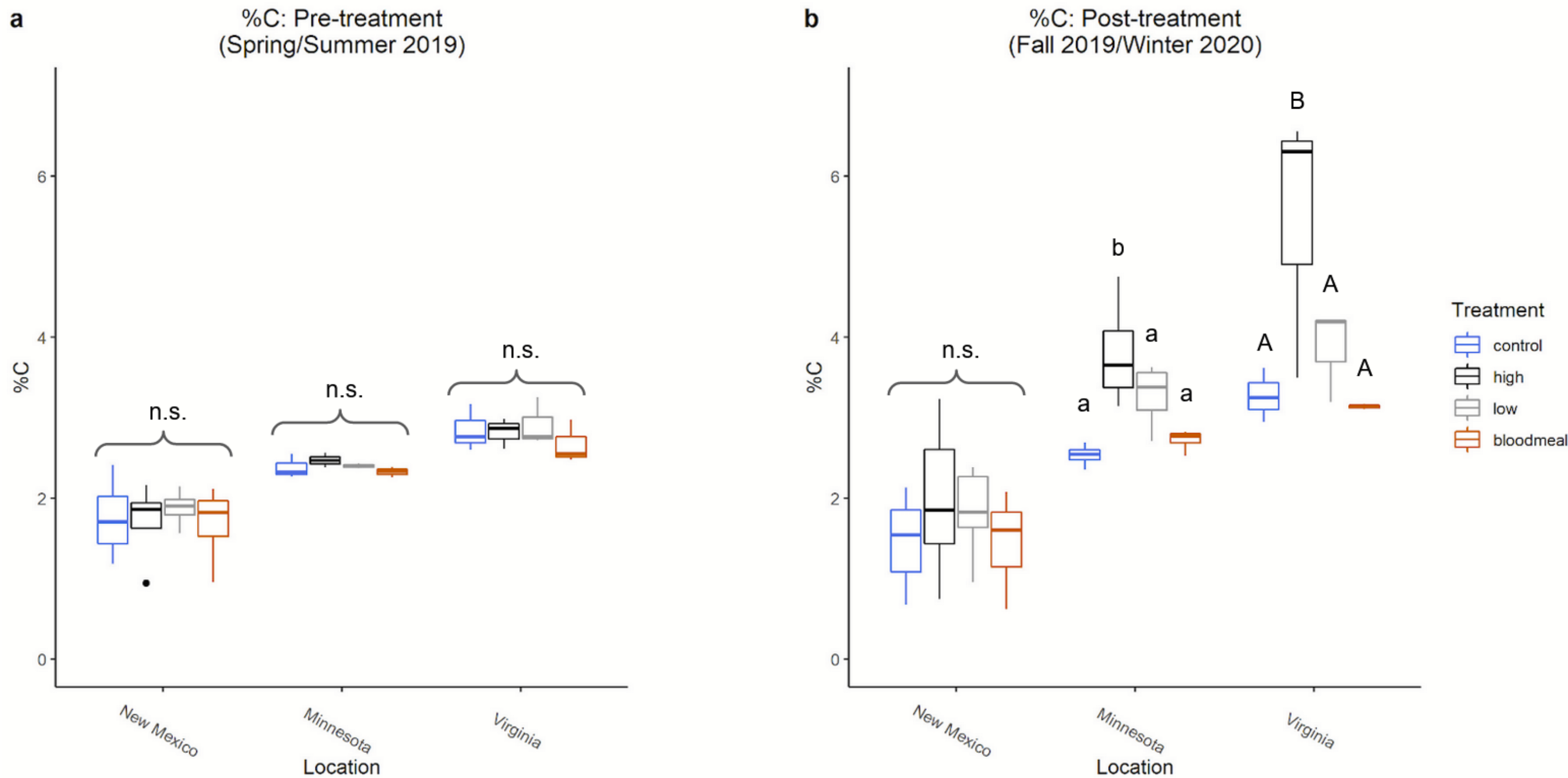

Figure 2.2 Total soil carbon values before and after biochar amendment. Pretreatment values (panel a) come from one round of sampling in the spring and summer of 2019. Post-treatment values (panel b) include data from samples taken from the fall/winter season at the end of 2019 into the beginning of 2020 and from the summer/fall of 2021. Letters indicate significant differences among treatments within one location and time point. Different letters indicate $p < 0.10$ in a post-hoc pairwise comparison (Tukey HSD test) of within-site means, n.s. = no significant difference between any two treatments. Capital letters are meant to distinguish between the two locations; pairwise comparisons are not being made between treatments across locations.

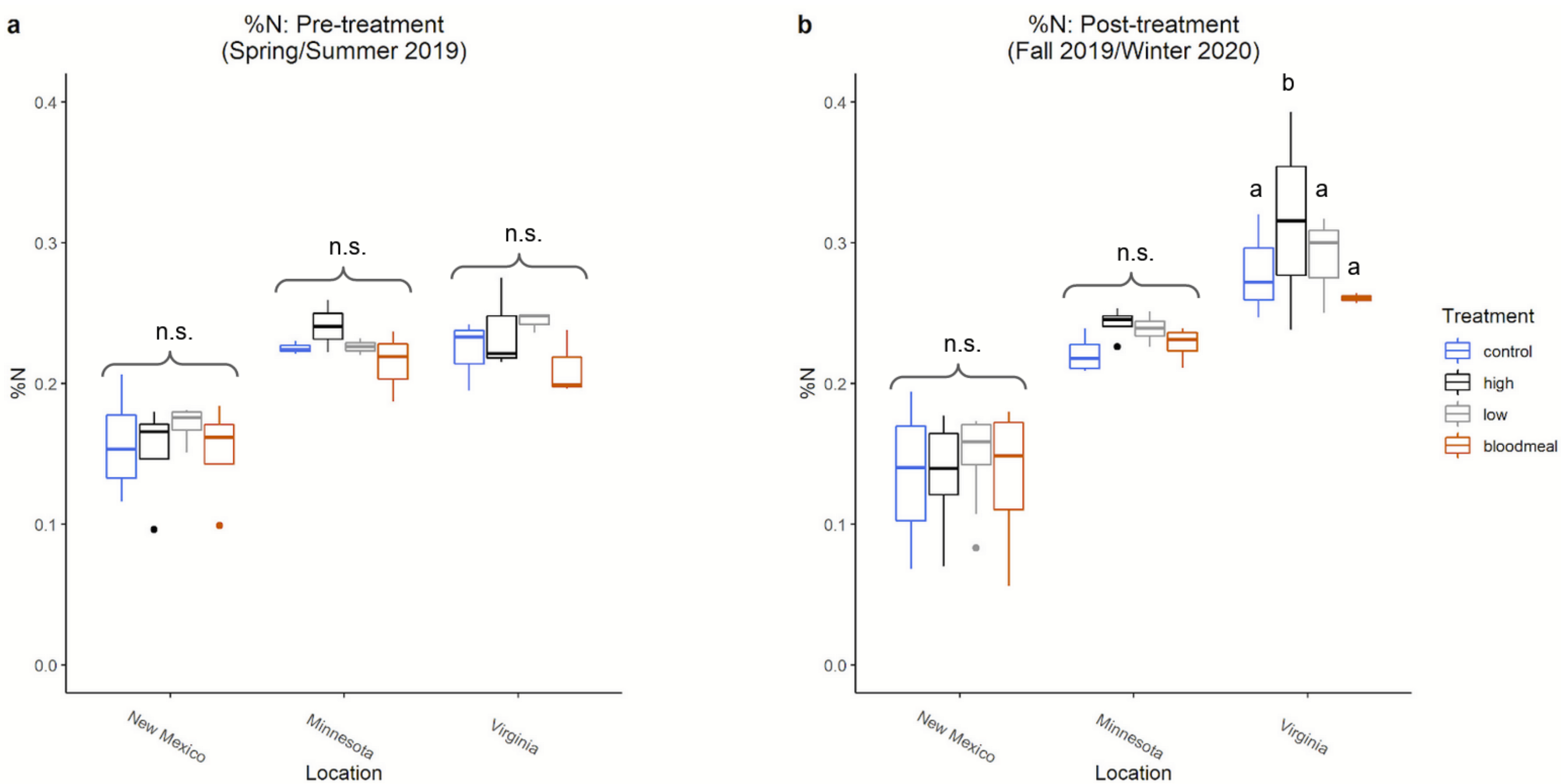

Figure 2.3 Total soil nitrogen values before and after biochar amendment. Pretreatment values (panel a) come from one round of sampling in the spring and summer of 2019. Post-treatment values (panel b) include data from samples taken from the fall/winter season at the end of 2019 into the beginning of 2020 and from the summer/fall of 2021. Letters indicate significant differences among treatments at one location and time point: different letters indicate $p < 0.10$ in a post-hoc pairwise comparison (Tukey HSD test) of within-site means, n.s. = no significant difference between any two treatments.

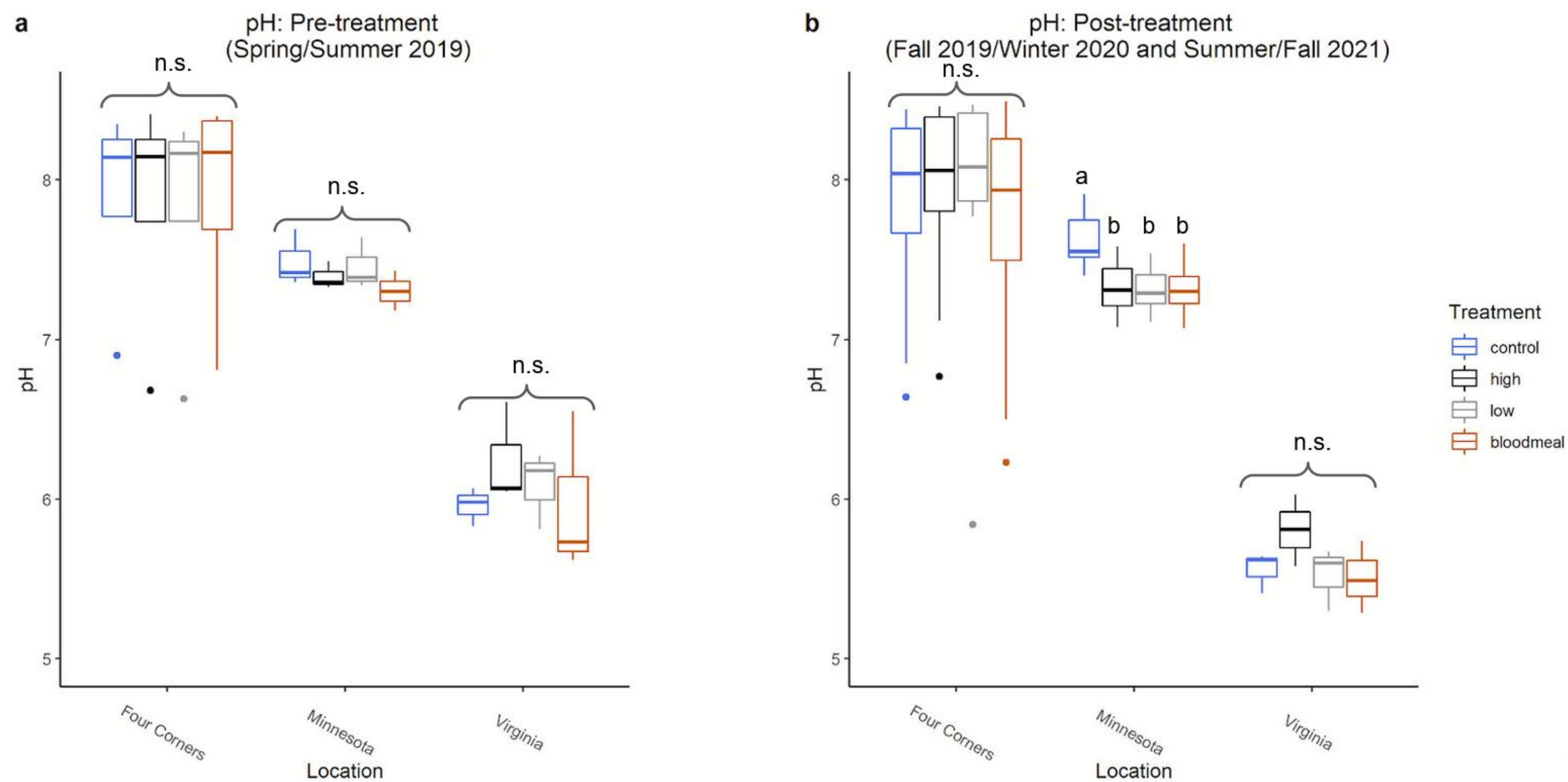

Figure 2.4 Soil pH values before and after biochar amendment. Pre-treatment values (panel a) come from one round of sampling in the spring and summer of 2019. Post-treatment values (panel b) include data from samples taken from the fall/winter season at the end of 2019 into the beginning of 2020 and from the summer/fall of 2021. Letters indicate significant differences among treatment s at one location and time point: different letters indicate $p < 0.10$ in a post-hoc pairwise comparison (Tukey HSD test) of within-site means, n.s. = no significant difference between any two treatments.

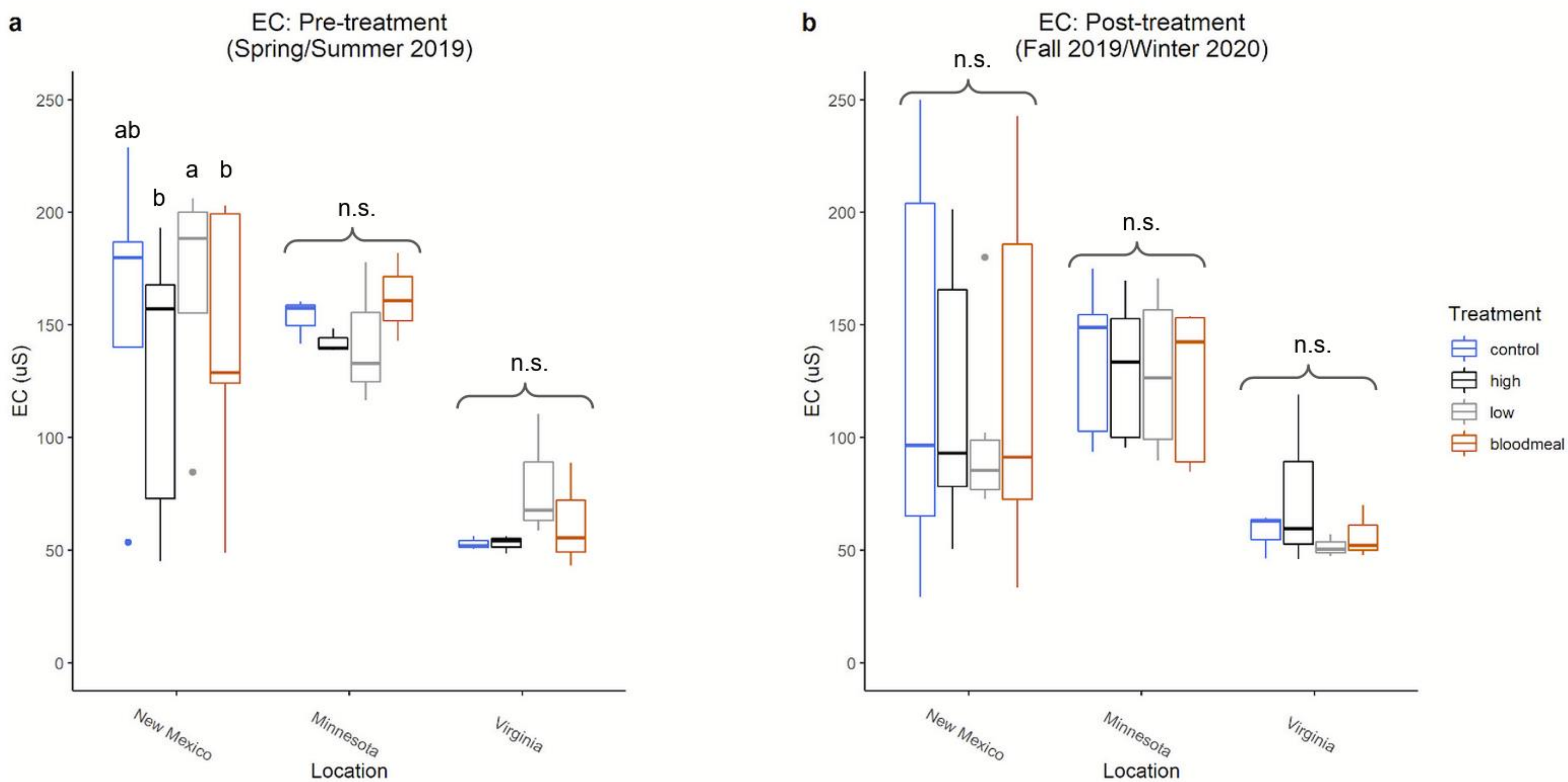

Figure 2.5 Electrical conductivity (EC) values before and after biochar amendment. Pre-treatment values (panel a) come from one round of sampling in the spring and summer of 2019. Post-treatment values (panel b) include data from samples taken from the fall/winter season at the end of 2019 into the beginning of 2020 and from the summer/fall of 2021. Letters indicate significant differences among treatments at one location and time point: different letters indicate $p < 0.10$ in a post-hoc pairwise comparison (Tukey HSD test) of within-site means, n.s. = no significant difference between any two treatments.

CHAPTER 3: Life cycle assessment of carbon balance for biochar made from three different feedstocks in Southwest Virginia pasture soil

Abstract

Biochar, a carbon-rich soil amendment produced by pyrolyzing organic feedstocks under low-oxygen conditions, is an appealing tool in the fight against climate change because it can be made relatively cheaply out of almost any organic waste material and offers both greenhouse gas emissions reduction and carbon sequestration when applied to soils. Some estimates suggest that applying biochar to agricultural soils worldwide could reduce net anthropogenic greenhouse gas emissions by up to 12%. In order to achieve these emissions reduction and carbon sequestration goals, it is crucial that greenhouse gasses emitted from biochar production and distribution do not cancel out any potential carbon storage benefit from the biochar itself. Life cycle assessment (LCA) is a method that quantifies the lifetime impact of a given product or process in units of energy, pollution, greenhouse gasses, etc., and can be used to help producers select the feedstock and pyrolysis method that results in the best carbon storage benefits in a given location. Here I conducted a life cycle assessment, paired with soil carbon data from experimental field plots, to determine whether small-scale biochar production and application to soil resulted in meaningful carbon sequestration in a pasture system in Southwest Virginia, and how three different local feedstocks (softwood, hardwood, and hay) compared in their carbon balance and global warming potential over the life cycle of the biochars. Feedstock, pyrolysis gas yield, transportation distance, and soil carbon retention all contributed to variation in the net carbon balance observed for each biochar system. Biochar made from softwood lumber scraps performed best overall, with the highest carbon storage and lowest global warming potential across a variety of scenarios, followed by biochar made from hardwood scraps. The hay biochar performed worst, with positive carbon emissions (i.e., more carbon released than stored over its life cycle) in most scenarios tested. These results suggest that producers can maximize the carbon storage capacity of biochar production and application by using waste products as feedstocks, investing in technologies to decrease or capture gas produced during pyrolysis, and minimizing transportation distances for feedstocks and biochar.

Introduction

Biochar, a carbon-rich material produced by pyrolyzing organic feedstocks under low-oxygen conditions, is an appealing tool in the fight against climate change for several reasons. It is relatively easy to make, is an opportunity to "up-cycle" agricultural waste products that would otherwise be discarded, and offers both greenhouse gas (GHG) emissions reduction and carbon dioxide removal, i.e., carbon sequestration (Groot et al. 2020, Lehmann et al. 2021). Woolf et al. (2010) estimated that biochar could be deployed at a global scale to reduce anthropogenic GHG emissions by 12% without negatively impacting human or environmental health; this equates to about 35% of all emissions associated with the global food system (Woolf et al. 2010, Crippa et al. 2021).

The carbon sequestration capacity of biochar stems from its increased resistance to decomposition in soil compared to unpyrolyzed feedstocks, decreased GHG emissions from soil due to negative priming of microbial carbon mineralization, increased plant productivity, and replacement of fossil fuels when energy produced during pyrolysis is captured and used for heating or electricity generation (Lehmann et al. 2021). Biochar's innate resistance to mineralization, leading to estimated residence times in the hundreds or thousands of years, comes from the highly aromatic chemical structure that develops during the pyrolysis process (Joseph et al. 2021). While raw organic material may only boost soil carbon in the short term, applying biochar to soil creates a long-lasting carbon pool. Biochar application can also drive non-additive sequestration, i.e., accumulation of soil carbon on top of the carbon that was added in the biochar itself. Non-additive sequestration is driven by interactions between biochar, soil organic matter, and microbes. Biochar can sorb organic compounds and form aggregates that protect organic matter from mineralization by microbes, or dilute the typical concentration of organic matter that is accessible to microbes (Joseph et al. 2021). These interactions reduce mineralization of soil carbon in a process called negative priming (Zimmerman et al. 2011).

While biochar, like any mitigation strategy, must be used worldwide to ameliorate the worst effects of climate change, the feedstocks and technologies used to produce it should be selected based on local availability and constraints. It is crucial that emissions from feedstock sourcing and shipping do not cancel out any carbon storage benefit from the production of the biochar itself. Life cycle assessment (LCA), which quantifies the lifetime impact of a given product or process in units of energy, pollution, greenhouse gasses, etc., can be used to help producers select the feedstock and pyrolysis method that results in the most carbon storage or economic benefits for a given application in a given location. Many LCAs have been published on biochar systems over the last 10+ years (see Matuštík et al. 2020). They are often done to predict the impact of building a new pyrolysis plant or diverting biomass from another use or waste stream. LCAs can help identify "hotspots" of greenhouse gas and carbon emissions within the life cycle and ensure that biochar systems live up to their potential as a carbon sequestration technology.

Here I present a life cycle assessment, paired with soil carbon data from experimental field plots, to determine whether biochar production and application to soil resulted in meaningful carbon sequestration in a pasture system in Southwest Virginia, and how three different local feedstocks (softwood, hardwood, and hay) compared in their carbon balance and global warming potential over the biochar's life cycle.

The goal of this life cycle assessment is to compare the total greenhouse gas emissions and global warming potential associated with the production, transport, and application to soil of biochar made from softwood, hardwood, and hay for an unimproved pasture agroecosystem in southwestern Virginia. The assessment was done using SimaPro 9.0 LCA software. Input and output data came from direct measurements and producer estimates, where possible, and the Ecoinvent database (Weidema et al. 2013). The objectives of the assessment are to determine:

a. Whether biochar production and use as a soil amendment has a net positive or negative impact on carbon storage in a southwest Virginia pasture system.
b. Which feedstock produces the largest negative impact (or smallest positive impact) on carbon emissions.
c. Which parameters in the life cycle have the greatest impact on the overall impact.

Methods

System Boundaries and Functional Unit

The functional unit for this assessment was 10 kg of biochar applied to agricultural soil at the Catawba Sustainability Center (CSC) in Catawba, VA (Conner 2022). The functional unit was chosen because it was the amount used in each of the experimental plots, so I could directly compare life cycle emissions to measured soil carbon results. The system boundaries included the harvest, transport, pyrolysis, and application of 10 kg of each biochar to plots at the CSC

(Figure 3.1). Biochar was produced at Sunrise Valley Farm in Hiwassee, VA. The production and transportation of the Exeter Charcoal Retort (Exeterra, LLC; Ohio, USA) itself is outside the scope of this analysis. Soil carbon content was analyzed two years after biochar amendment and estimates for longer-term carbon sequestration were estimated from literature values (Spokas 2010). The focus for this analysis was on carbon emissions and sequestration. Other studies at the experimental site have looked into the effects of biochar on other soil properties; those will not be discussed here.

Life Cycle Inventory Analysis

A life cycle inventory analysis and impact assessment was performed in SimaPro, a widely-used life cycle assessment software developed by PRé Sustainability. SimaPro allows the user to construct processes and life cycles for products and compare different scenarios for raw material sourcing, manufacturing, transportation, use, and disposal or recycling. SimaPro draws from databases of materials and processes published by research and consulting organizations around the world. Life cycle inventory analyses report the impact of a products' life cycle in units of energy, greenhouse gas emissions, pollutants, land use change, or other outputs that are relevant to environmental and human health.

For my inventory analysis, I pulled data from producer measurements and estimates, literature values, and Ecoinvent database processes (Weidema et al. 2013) to quantify the amount and forms of greenhouse gasses emitted from feedstock harvest to biochar application (Table 3.9). The total amount of carbon released was also calculated from these values to allow for a direct comparison between the mass of carbon emitted and the mass of carbon stored in soil for each feedstock.

I. *Feedstock collection and transportation*

All feedstocks were sourced locally to minimize transportation distances for feedstock and biochar. The hardwood and softwood biochars were made from a mix of lumber scraps and slash left over after harvest and processing at Sunrise Valley Farm in Hiwassee, VA, where the biochar was also produced. The hardwood was primarily oak, hickory, and maple, and the softwood was primarily Virginia pine. The hardwood and softwood LCAs begin with the transportation of scrap wood from the mill site to the retort; emissions associated with the lumber processing that produced the scraps is not accounted for here, as the feedstocks were waste products from an unrelated process. The hardwood and softwood feedstocks traveled 0.3km (0.2 miles) from the mill to the retort. The hay feedstock was a mix of warm season grasses grown at the Catawba Sustainability Center, where all three biochars were later applied to experimental soil plots. The hay biochar LCA includes all emissions associated with harvest, as the hay was grown for the purpose of biochar production. The hay traveled 64km (40 miles) from where it was grown to the retort site in a standard pickup truck (Ford F150). Travel distances were scaled to the fraction of a truckload (estimated at ~400 kg based on producer experience) being transported to account for differences in feedstock mass.

II. Biochar Production and Application

The feedstocks were pyrolyzed using an Exeter Charcoal Retort (Exeterra, LLC; Ohio, USA). This was a slow pyrolysis method maximized for biochar production, with feedstocks held at a maximum temperature of ~500℃ for 1-2 hours. Apart from the internal recycling of liquid and gaseous co-products to fuel pyrolysis, no co-products (syngas or bio-oil) were captured to be used as fuel. See Table 3.1 for details on pyrolysis time and temperature, Table 3.2 for pyrolysis product yield, and Tables 3.3 - 3.8 for the data used to calculate pyrolysis yields for all feedstocks. Both the relative solid, liquid, and gas yields and the composition of the gas fraction were estimated from Brownsort 2009; the gas composition was estimated as follows: 32% CO_2, 34% CO, 24% H, and 10% CH_4 by volume (Brownsort 2009).

After pyrolysis the biochar was transported in a pickup truck and applied to 2x2m plots established in a warm season grass pasture at the Catawba Sustainability Center. Biochar was applied by hand at a rate of 2.5 kg/m^2 and raked evenly across plots on 27 September 2019. All plots were disced on 11 November 2019 to ensure the biochar was well incorporated into the soil.

Life Cycle Impact Assessment

Life cycle impact assessment is an additional step that translates the raw outputs calculated in the life cycle inventory analysis into environmental or human health outcomes like global warming, aquatic toxicity, or resource depletion. I used the International Panel on Climate Change 100-year Global Warming Potential (IPCC GWP 100yr) impact assessment method to focus on the long-term impact of greenhouse gas emissions produced over the life cycle of the biochar (IPCC 2013). GWP results are reported in units of CO_2 equivalences (kg CO_{2eq}), or the amount of warming associated with each greenhouse gas over 100 years scaled to units of CO_2.

Soil Carbon

One year after biochar application, three samples were taken from the top 10cm of soil in each experimental plot, homogenized, passed through a 2mm sieve to remove larger rocks and plant materials, and air dried. Subsamples were then ground using a SPEX SamplePrep 8000M Mixer/Mill (SPEX SamplePrep, Metuchen, New Jersey, USA) and run on an Elementar vario MAX cube CHN analyzer (Elementar Americas, Inc., Ronkonkoma, New York, USA) to measure organic carbon content. The percentage of biochar carbon expected to remain long-term (100+ years) in the soil was estimated based on the volatile matter content of the biochar (Spokas 2010). A separate subsample from each experimental plot was separated into five size fractions (2-4mm, 250µm - 2mm, 53µm - 250µm, and <53µm) using the wet sieving method described in Six et al (1998). These size fractions were oven dried at 105°C, then ground and analyzed for CN content like the unfractionated samples to determine where carbon accumulated after biochar application. All soil carbon values reported here are increases above the pre-treatment time point for the same plots, not absolute carbon content.

Sensitivity Analysis for Gas Yield and Transportation Distance

Pyrolysis was the step in the life cycle that contributed the most to greenhouse gasses and global warming potential for all three biochars, followed by fossil fuel emissions from transportation and harvest, in the case of hay (see Tables 3.10 and 3.11). A sensitivity analysis was performed to determine how overall carbon balance and global warming potential would change if the relative yield of the gas co-product during pyrolysis were higher or lower than the estimates I used in the primary analysis, and under three different transportation scenarios.

For the original analysis, liquid and gas co-product percent yields by mass were estimated from the midpoint of ranges for slow pyrolysis in Brownsort (2009). While the mass yields of biochar from these pyrolysis runs were known, I had no direct measures of the liquid or gas co-products. It is possible that the relative size of the gas fraction, which contributed significantly to overall greenhouse gas emissions, was well above or below the estimate used. To test the impact of different gas yields on overall emissions and global warming potential, I re-ran the life cycle assessment under two other scenarios: low gas yield (0.5x the original estimate by mass) and high gas yield (1.5x the original estimate by mass) (Table 3.12).

Three transportation scenarios were also analyzed: low transportation, where all feedstocks were harvested, pyrolyzed, and applied on-site (5 km total, half as feedstock and half as biochar); medium transportation, where all feedstocks and biochar traveled the same distance as hay in the original analysis (128 km total, half as feedstock and half as biochar); and high transportation, where all feedstocks traveled double the original distance for hay (256 km total, half as feedstock and half as biochar) (Table 3.13). These distances represent a range of hyper-local to regional-scale biochar production and application systems. As with the original calculation, travel distances were scaled to the fraction of a truckload (400 kg) needed to transport feedstock and biochar per 10 kg biochar.

The mass of carbon emitted under each of the gas yield and transportation scenarios was compared to three levels of soil carbon residence time over 100 years (no loss, 10% loss, and 50% loss). The 0%, 10% and 50% loss values were chosen as they represent the spectrum of estimated half life values for biochar with similar volatile matter profiles as ours: 24%, 26%, and 28% by mass, respectively, for softwood, hardwood, and hay (Spokas 2010). See Table 3.15 for results.

Results and Discussion

Life Cycle Inventory Analysis

The primary emissions relevant to the carbon balance across each biochar's life cycle were carbon dioxide, carbon monoxide, and methane (Table 3.10 and Figure 3.3). Emissions from the pyrolysis process itself represented the largest source of each of these gasses, followed by emissions from transportation and harvest (Table 3.10 and Figure 3.3). The hay biochar emitted the most carbon over the course of its life cycle, with 12.16 kg emitted per 10 kg biochar produced. The emissions comprised 26.6 kg CO_2 (7.2 kg carbon), 9.0 kg CO (3.8 kg carbon), and 1.5 kg methane (1.13 kg carbon). The hardwood biochar emitted 4.48 kg carbon per 10 kg biochar, from 14.8 kg CO_2 (2.0 kg carbon), 4.5 kg CO (1.9 kg carbon), and 0.75 kg CH_4 (0.56 kg carbon). The softwood emitted the least total carbon, at 2.71 kg per 10 kg biochar. The emissions were 9.2 kg CO_2 (1.3 kg carbon), 2.6 kg CO (1.1 kg carbon), and 0.44 kg methane (0.33). For a full breakdown of emissions see Table 3.10.

Life Cycle Impact Assessment

The global warming potential values followed the same pattern as the total carbon emitted (Figure 3.4). The hay biochar life cycle had the greatest 100-year global warming potential, at 81.95 kg CO_2 eq, followed by hardwood with 30.3 kg CO_2 eq and softwood with 18.1 kg CO_2 eq per 10 kg of biochar produced (Table 3.11 and Figure 3.4). Methane emissions, particularly from pyrolysis, were the main driver for overall GWP, contributing over 50% to the overall warming potential value for each feedstock system despite making up only ~4% of greenhouse gas emissions by mass. The 100-year time scale used for this impact assessment is relevant to the results as well. In the short term the relative impact of methane emissions on warming is even more significant, as it is a short-lived but powerful greenhouse gas compared to carbon dioxide (IPCC 2013).

The majority of the global warming potential associated with each feedstock was driven by emissions during pyrolysis (Table 3.11). For softwood and hardwood, methane emissions from pyrolysis (13.4 and 22.9 kg CO_2 eq, respectively) made up ~75% of total GWP. Fossil fuel emissions from transportation of feedstock and biochar only contributed about 4% of total GWP. Harvest and transportation emissions drove a larger proportion of overall GWP for the hay feedstock, at 28%. In the hay system, CO_2 emissions from fossil fuels (13.3 kg) were greater than CO_2 emissions from pyrolysis (13.15 kg), and nitrous oxide contributed ~12% of total GWP (9.5 kg CO_2 eq). Nitrous oxide emissions represented less than 0.005 kg CO_2 eq for the two wood feedstocks. These results demonstrate the importance of choosing feedstocks that require minimal on-farm processing (like cutting and baling, in the case of hay) and minimal transportation to the pyrolysis and soil application sites.

Soil Carbon

One year after biochar amendment, soil carbon content had increased in all biochar-amended plots compared to their pre-amendment soil carbon levels. The softwood biochar plots had the smallest increase, with an average of 1.82 kg/m^2 more carbon after biochar amendment than before. The hardwood biochar plots increased by 2.11 kg/m^2. The hay biochar plots had the largest increase, with 3.01 kg/m^2 more carbon than before amendment. These increases equated to 7.29, 8.42, and 12.02 kg carbon, respectively, per 10 kg of softwood, hardwood, or hay biochar applied (Table 3.14, Figures 3.2 and 3.3).

The soil size fraction results provided insights on where carbon accumulated after biochar application to soil (Figure 3.3). The 250µ - 2mm fraction (generally considered "macroaggregates") had the largest increase in soil carbon for each of the feedstocks. Hay biochar increased the carbon content of this fraction by 6.6 kg per 10 kg biochar, hardwood increased by 3.8 kg, and softwood increased by 4.8 kg. Hardwood had a similar increase of 3.4 kg in the 2 - 4mm fraction, which was primarily made up of larger pieces of biochar. Softwood biochar created a roughly equal increase in the 2 - 4mm and the 53µ - 250µ (microaggregate) fractions (1.2 and 1.1 kg, respectively), as did hay biochar (2.6 kg and 2.7 kg, respectively). Hardwood biochar increased the carbon content of the 53µ - 250µ fraction by 1.2 kg. The carbon increase in the >53µ (organo-mineral aggregates) fraction was the smallest across all feedstocks (0.1 kg for hay, 0.05 kg for hardwood, and 0.2 kg for softwood). Softwood showed the largest increase in that fraction both as a percentage of its total soil carbon increase and across the three feedstocks.

Net Carbon Balance over the Life Cycle

While hay biochar produced the largest increase in soil carbon, it also produced the most carbon emissions, creating a net positive carbon balance (i.e., more carbon was released than stored) over its life cycle. The hay biochar system produced net emissions of 0.14 kg carbon per 10 kg of biochar. The softwood and hardwood biochar systems both had net storage of carbon over their life cycles, with net carbon balances of -4.58 kg and -3.94 kg, respectively, per 10 kg of biochar (Figure 3.2).

Sensitivity Analysis of for Gas Yield and Transportation Distance

The net carbon storage of the softwood biochar system was robust to all but three of the most extreme scenarios tested in the sensitivity analysis. Under the 100% and 90% soil carbon retention scenarios softwood biochar produced net carbon storage regardless of transportation distance or pyrolysis gas yield. Only at the 50% soil carbon retention level and under the medium and high transportation and high gas yield scenarios did the softwood biochar system emit more carbon than it stored.

The hardwood biochar system also performed well in most scenarios, but the net carbon storage observed in the original analysis was less robust to other transportation and pyrolysis gas yield scenarios. At the 100% and 90% soil carbon retention levels, the hardwood biochar system achieved net carbon storage in all cases except the high transportation scenario, where the system emitted 0.04 kg carbon per 10 kg biochar. At the 50% soil carbon retention level, the hardwood biochar system only had a net negative carbon balance under the low gas yield scenario.

The hay biochar system performed worst in the original analysis and under the alternate scenarios examined in the sensitivity analysis. There was only net carbon storage at the 100% and 90% soil carbon retention levels under the low transportation and low gas yield scenarios. In all other scenarios, more carbon was emitted than stored.

Impact of Feedstock Choice on Carbon Balance

In our analysis, softwood and hardwood outperformed hay in carbon sequestration over the life cycle of biochar feedstocks and their application to soils. The wood feedstocks' density and higher lignin content made for an efficient conversion to biochar (Groot et al. 2020, Lehmann et al. 2006), producing more biochar per unit feedstock and requiring fewer pyrolysis runs than the hay to create the same mass of biochar. The scrap wood and lumber also had the benefit of being waste materials left over from lumber processing and up-cycled, rather than produced to purpose, i.e., the energy associated with felling, cutting, and milling wood feedstock is not accounted for here in contrast to warm season grass hay which is often produced intentionally for bioenergy. Several LCAs have concluded that waste product feedstocks are best for creating net negative emissions in biochar systems, while biochar systems that use feedstocks, particularly grasses, grown specifically for pyrolysis tend to result in neutral-to-positive emissions (Ibarrola et al. 2012, Hammond et al. 2011, Roberts et al. 2010). The most appropriate feedstock for a biochar system in a given area will vary depending on what is locally available and what the alternative uses of the biomass would be (see, for example, Azzi et al. 2019 and Puettman et al. 2017). Cost of feedstocks and shipping can also be a limiting factor, yet another reason why it is best to use waste products sourced as close as possible to the pyrolysis and application sites (Groot et al. 2020).

Pyrolysis Technology and Carbon Balance

The majority of emissions associated with each biochar came from the pyrolysis process itself. This aligns with the results from many other LCAs conducted on biochar systems (see

Matuštík et al. 2020). Given the outsized impact of pyrolysis emissions on the overall carbon balance, making adjustments to maximize biochar production and limit greenhouse gas emissions during pyrolysis would be a key way to improve the carbon storage of the system as a whole. Pyrolysis temperature and timing are major drivers of biochar mass yield; slow heating rate, long residence time, and high temperatures (~500-1000°C) tend to produce the most biochar per unit feedstock while also creating a product with desirable physical and chemical characteristics (Joseph et al. 2021, Weber and Quicker 2018, Lehmann et al. 2006).

Many life cycle assessments performed for biochar systems have focused on larger-scale pyrolysis plants where co-production of energy (and therefore avoided emissions from fossil fuels) contribute to overall carbon sequestration (Azzi et al. 2019, Homagain et al. 2015, Hammond et al. 2011). Here I focused solely on the carbon sequestration potential of biochar applied to soil. Many smaller-scale biochar producers do not have the technology to capture and use co-products for energy, so there must be carbon offset benefits from application to soil alone in order for biochar to appeal to small and medium-sized diversified farms. Even so, I saw net carbon storage driven by increases to soil carbon alone with our hardwood and softwood biochars in most scenarios, which is supported by results from other LCAs where soil carbon storage is the primary contribution to overall carbon sequestration (Matuštík et al. 2020). These results demonstrate that local-level biochar production can play a part in carbon sequestration efforts even when advanced pyrolysis technology is not available or practical due to cost or scale.

Soil Carbon Dynamics

While all three types of biochar drove increases in soil carbon, hay was notable in that the soil carbon increase was larger than the amount of biochar added (increase of 12.02 kg carbon per plot after 10 kg of biochar was applied). Hay biochar may have induced a negative priming effect, meaning that hay biochar amendment slowed the mineralization and/or increased stabilization of plant- and microbially-derived soil organic matter and resulted in soil carbon content that exceeded the carbon added as biochar. A study by Zimmerman et al. (2011) that compared the priming effects of oak, pine, and grass biochar produced at a range of pyrolysis temperatures over 500+ days post-application showed that biochar made from grass suppressed soil organic carbon mineralization across the board. For their hardwood and softwood biochars, the pyrolysis temperature influenced whether soil carbon mineralization was elevated or suppressed, with a correlation between pyrolysis temperature and suppression of mineralization (Zimmerman et al. 2011). Most biochars demonstrated some level of suppression on soil carbon mineralization later in the incubation, suggesting that biochar's negative priming capacity increases over time (Zimmerman et al. 2011). Blanco-Canqui et al (2019) also found evidence of negative priming in a long-term field experiment, where plots amended with wood biochar exhibited an increase in soil carbon that was almost double the biochar carbon added six years earlier. More long-term studies are needed to clarify soil carbon dynamics after biochar application, as results from short-term or lab-based experiments may not reflect results under actual agricultural conditions (Vijay et al. 2021, Schmidt et al. 2021, Maestrini et al. 2015).

Accurately estimating soil carbon residence over the long term (in this study, 100 years) is critical to predicting the net carbon balance of biochar systems. Due to the range of feedstocks and pyrolysis methods included under the umbrella term of "biochar", coupled with the difficulty of estimating how biochar will behave over hundreds to thousands of years, literature values for biochar carbon residence time span orders of magnitude (Gurwick et al. 2013). In general, biochars produced at higher pyrolysis temperatures are expected to mineralize more slowly over time (Joseph et al. 2021, Fang et al. 2014).

The size fraction where each biochar contributed the most carbon could provide additional insight into the longevity of the additional soil carbon observed after biochar application. Several studies have demonstrated increased soil aggregation following biochar application, as well as preferential incorporation of biochar carbon in microaggregates and the organo-mineral fraction, generally defined as <250μm in size (Yoo et al. 2017, Weng et al. 2017). Organic matter protected within aggregates or stabilized on clay particles in these size fractions is generally believed to be well-protected from microbial access and mineralization, and therefore represent long-term pools of soil carbon (Six et al. 2002). In our soil samples collected one year after biochar application, the soil carbon gains in the <250μm fractions represented 23% (hay), 17% (softwood) and 15% (hardwood) of total increases in soil carbon. These percentages may be expected to increase as physical and biological processes break down the larger pieces of biochar. More long-term biochar field studies are needed to examine how biochar is incorporated into and/or accelerates the formation of soil aggregates over decades to centuries.

Conclusion

This life cycle assessment of biochar made from up-cycled softwood and hardwood waste, and purpose-grown hay applied to Southwest Virginia pasture soils demonstrated that local-scale biochar production and use can create meaningful increases in soil carbon and overall carbon sequestration. The differences in carbon balance and global warming potential over the full life cycles of the three feedstocks I investigated made it clear that not all biochar is equal with regard to its carbon storage capacity. The hay biochar, which contributed the most to soil carbon, was also the most carbon-intensive to harvest and transport. Over its life cycle it emitted 0.14 kg more carbon than it stored, and was only predicted to store carbon under a few sensitivity analysis scenarios that assumed minimal soil carbon loss, transportation distance, and gas yield during pyrolysis. In contrast, the softwood and hardwood feedstocks contributed somewhat less to increases in soil carbon storage, but achieved negative carbon balances over their life cycles (-4.58 kg and -3.94 kg, respectively), due primarily to the lower emissions associated with their harvest, pyrolysis, and transportation. The carbon balance of upcycled-wood pyrolysis remained negative in all but a few sensitivity analysis scenarios that assumed 50% carbon loss and higher transportation distance and pyrolysis gas yields. These results suggest that producers can maximize the carbon storage capacity of biochar systems by using agricultural or timber processing waste products as feedstocks, investing in appropriately-scaled technologies to decrease or capture gas produced during pyrolysis, and minimizing transportation distances for feedstocks and biochar.

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Tables

Table 3.1 Pyrolysis information for each biochar. All biochar was produced using an Exeter Charcoal Retort (Exeterra, LLC; Ohio, USA).

Feedstock	Pyrolysis Method	Approx. Total Time (hrs)	Approx. Time at Max Temp (hrs)	Approx. Max Temp (°C)
Hardwood	Charcoal retort	9 – 9.5	2	500
Softwood	Charcoal retort	10	1.6 – 2	500
Hay	Charcoal retort	4 – 6.5	1 – 2.5	450 – 500

Table 3.2 Solid, liquid, and gas pyrolysis yield by mass. Yield of solid fraction (biochar) was directly measured by the producer. Liquid and gas fraction yields are estimated from Brownsort 2009 for slow pyrolysis of similar feedstocks. All yields were calculated as percent of dry feedstock.

Feedstock	Solid Products (Biochar) Yield by Mass	Liquid Products (Bio-oil) Yield by Mass	Gaseous Products (Syngas) Yield by Mass
Hardwood	29%	35.5%	35.5%
Softwood	28%	52%	20%
Hay	10%	65%	25%

Table 3.3 Hardwood feedstock to biochar conversion data.

Name	Value	Source	Notes
Feeder Wood Volume per Run	0.25 m^3	Producer estimate	Wood burned in the outside ring of the retort to heat interior
Feeder Wood Mass per Run	200 kg	Feeder Wood Volume per Run (0.25 m^3) * Estimated Feeder Wood Density (800 kg/m^3)	Feeder wood was additional hardwood scraps
Retort Capacity	1.70 m^3	Measurement of retort	
Packed Retort Air Space	20%	Producer estimate	
Wood Density	600-930 kg/m^3	Producer estimate	Mix of oak, hickory, and maple
Feedstock Volume per Run	1.36 m^3	Retort Capacity (1.70m^3) * Non-Air Space (0.8)	
Feedstock Mass per Run	816-1265 kg	Feedstock Volume per Run (1.36m^3) * Wood Density (600-930 kg/m^3)	
Volumetric Yield (Feedstock -> Biochar)	60%	Producer estimate	
Biochar Bulk Density	295 kg/m^3	Biochar lab analysis	Converted from 18.4lb/ft^3
Biochar Yield per Run by Volume	0.82 m^3	Feedstock Volume (1.36m^3) * Volumetric Yield (0.6)	
Biochar Yield per Run by Mass	242 kg	Biochar Bulk Density (295 kg/m^3) * Biochar Yield per Run by Volume (0.82m^3)	
Fraction of Run Needed for 10 kg of Biochar	0.041	$\frac{10\ kg\ biochar}{1} \times \frac{1\ run}{242\ kg\ biochar}$	
Feeder Wood per 10 kg Biochar	8.2 kg	Feeder Wood Mass per Run (200 kg) * Fraction of Run Needed for 10 kg of Biochar (.041)	
Feedstock Volume per 10 kg Biochar	0.054 m^3	Fraction of Run Needed for 10 kg of Biochar (0.041) * Feedstock Volume per Run (1.36 m^3)	
Feedstock Mass per 10 kg Biochar	33.46 - 51.87 kg	Fraction of Run Needed for 10 kg of Biochar (0.041) * Feedstock Mass per Run (816 - 1265 kg)	

Table 3.4 Hardwood pyrolysis yield data

Name	Value	Source	Notes
Feedstock Mass per 10 kg Biochar	42.7 kg	Midpoint of range calculated in Table 3.3	
Feedstock Moisture Content by Mass	20%	Biochar producer estimate	
Feedstock Water Content Mass	8.5 kg	Feedstock Mass per 10 kg Biochar (42.7) * Feedstock Moisture Content by Mass (20%)	Assumed to be lost as water vapor during pyrolysis
Feedstock Dry Mass	34.2 kg	Feedstock Mass per 10 kg Biochar (42.7 kg) - Feedstock Water Content Mass (8.5 kg)	
Biochar Yield by Mass	29%	10 kg Biochar / Feedstock Dry Mass (34.2)	
Liquid Yield by Mass	35.5%	[100% - Biochar Yield by Mass (29%)] / 2	Falls within range from Brownsort 2009
Gas Yield by Mass	35.5%	[100% - Biochar Yield by Mass (29%)] / 2	Falls within range from Brownsort 2009

Table 3.5 Softwood feedstock to biochar conversion data

Name	Value	Source	Notes
Feeder Wood Volume per Run	0.25 m^3	Producer estimate	Wood burned in the outside ring of the retort to heat interior
Feeder Wood Mass per Run	200 kg	Feeder Wood Volume per Run (0.25 m^3) * Estimated Feeder Wood Density (800 kg/m^3)	Feeder wood was additional hardwood scraps
Retort Capacity	1.70 m^3	Measurement of retort	
Packed Retort Air Space	20%	Producer estimate	
Wood Density	350-500 kg/m^3	Producer estimate	White pine
Feedstock Volume per Run	1.36 m^3	Retort Capacity (1.70m^3) * Non-Air Space (0.8)	
Feedstock Mass per Run	476 - 680 kg	Feedstock Volume per Run (1.36m^3) * Wood Density (350-500 kg/m^3)	
Volumetric Yield (Feedstock -> Biochar)	50%	Producer estimate	
Biochar Bulk Density	190.6 kg/m^3	Biochar lab analysis	Converted from 11.9lb/ft^3
Biochar Yield per Run by Volume	0.68 m^3	Feedstock Volume (1.36m^3) * Volumetric Yield (0.5)	
Biochar Yield per Run by Mass	129.6 kg	Biochar Bulk Density (190.6 kg/m^3) * Biochar Yield per Run by Volume (0.68m^3)	

Name	Value	Source	Notes
Feeder Wood Volume per Run	$0.25\ m^3$	Producer estimate	Wood burned in the outside ring of the retort to heat interior
Feeder Wood Mass per Run	200 kg	Feeder Wood Volume per Run ($0.25\ m^3$) * Estimated Feeder Wood Density ($800\ kg/m^3$)	Feeder wood was additional hardwood scraps
Fraction of Run Needed for 10kg of Biochar	0.077	$\frac{10\ kg\ biochar}{1} \text{ x } \frac{1\ run}{129.6\ kg\ biochar}$	
Feeder Wood per 10kg Biochar	15.4 kg	Feeder Wood Mass per Run (200 kg) * Fraction of Run Needed for 10 kg of Biochar (.077)	
Feedstock Volume per 10kg Biochar	$0.10\ m^3$	Fraction of Run Needed for 10 kg of Biochar (0.077) * Feedstock Volume per Run ($1.36\ m^3$)	
Feedstock Mass per 10kg Biochar	36.7 - 52.4 kg	Fraction of Run Needed for 10 kg of Biochar (0.077) * Feedstock Mass per Run (476-680 kg)	

Table 3.6 Softwood pyrolysis yield data

Name	Value	Source	Notes
Feedstock Mass per 10 kg Biochar	44.6 kg	Midpoint of range calculated in Table 3.5	
Feedstock Moisture Content by Mass	20%	Biochar producer estimate	
Feedstock Water Content Mass	8.9 kg	Feedstock Mass per 10 kg Biochar (44.6) * Feedstock Moisture Content by Mass (20%)	Assumed to be lost as water vapor during pyrolysis
Feedstock Dry Mass	35.7 kg	Feedstock Mass per 10 kg Biochar (44.6 kg) - Feedstock Water Content Mass (8.9 kg)	
Biochar Yield by Mass	28%	10 kg Biochar / Feedstock Dry Mass (35.7 kg)	
Liquid Yield by Mass	52%	Estimate based on range in Brownsort 2009	
Gas Yield by Mass	20%	Estimate based on range in Brownsort 2009	

Table 3.7 Hay feedstock to biochar conversion data

Name	Value	Source	Notes
Feeder Wood Volume per Run	0.25 m^3	Producer estimate	Wood burned in the outside ring of the retort to heat interior
Feeder Wood Mass per Run	200 kg	Feeder Wood Volume per Run (0.25 m^3) * Estimated Feeder Wood Density (800 kg/m^3)	Feeder wood was additional hardwood scraps
Retort Capacity	1.70 m^3	Measurement of retort	
Packed Retort Air Space	20%	Producer estimate	
Hay Density	163 kg DM/m^3	Shinners et al. 2010	Warm season grasses
Feedstock Volume per Run	1.36 m^3	Retort Capacity (1.70m^3) * Non-Air Space (80%)	
Feedstock Mass per Run	222 kg	Feedstock Volume per Run (1.36 m^3) * Hay Density (163 kg/m^3)	
Volumetric Yield (Feedstock -> Biochar)	10%	Producer estimate	
Biochar Bulk Density	135 kg/m^3	Biochar lab analysis	Converted from lb/ft^3
Biochar Yield per Run by Volume	0.136 m^3	Feedstock Volume (1.36 m^3) * Volumetric Yield (0.1)	
Biochar Yield per Run by Mass	18.36 kg	Biochar Bulk Density (135 kg/m^3) * Biochar Yield per Run by Volume (0.136 m^3)	
Fraction of Run Needed for 10 kg of Biochar	0.54	$\frac{10\ kg\ biochar}{1} \times \frac{1\ run}{18.36\ kg\ biochar}$	

Name	Value	Source	Notes
Feeder Wood Volume per Run	0.25 m^3	Producer estimate	Wood burned in the outside ring of the retort to heat interior
Feeder Wood Mass per Run	200 kg	Feeder Wood Volume per Run (0.25 m^3) * Estimated Feeder Wood Density (800 kg/m^3)	Feeder wood was additional hardwood scraps
Feeder Wood per 10 kg Biochar	108 kg	Feeder Wood Mass per Run (200 kg) * Fraction of Run Needed for 10 kg of Biochar (0.54)	
Feedstock Volume per 10 kg Biochar	0.73 m^3	Fraction of Run Needed for 10 kg of Biochar (0.54) * Feedstock Volume per Run (1.36 m^3)	
Feedstock Mass per 10 kg Biochar	119.8 kg	Fraction of Run Needed for 10 kg of Biochar (0.54) * Feedstock Mass per Run (222 kg)	

Table 3.8 Hay pyrolysis yield data

Name	Value	Source	Notes
Feedstock Mass per 10 kg Biochar	119.8 kg	Midpoint of range calculated in Table 3.7	
Feedstock Moisture Content by Mass	20%	Biochar producer estimate	
Feedstock Water Content Mass	24 kg	Feedstock Mass per 10 kg Biochar (119.8) * Feedstock Moisture Content by Mass (20%)	Assumed to be lost as water vapor during pyrolysis
Feedstock Dry Mass	95.8 kg	Feedstock Mass per 10 kg Biochar (119.8 kg) - Feedstock Water Content Mass (8.9 kg)	
Biochar Yield by Mass	10%	10 kg Biochar / Feedstock Dry Mass (95.8 kg)	
Liquid Yield by Mass	65%	Estimate based on range in Brownsort 2009	
Gas Yield by Mass	25%	Estimate based on range in Brownsort 2009	

Table 3.9 Ecoinvent processes used for the life cycle assessment

Process	Unit	Amount	Notes
Wood and wood waste, 9.5 MJ per kg	kg	Determined by volume and density of hardwood and softwood used	
Hay, organic, intensive {RoW}| production | APOS, U [ADAPTED]	kg	119.8	Ecoinvent process was adapted to better represent our system; processes below were included, all others were removed.
- Mowing, by rotary mower {GLO}| market for | APOS, U	ha	0.0005	
- Swath, by rotary windower {GLO}| market for | APOS, U	ha	0.0005	
- Baling {RoW}| processing | APOS, U	p	0.004	
Transport, passenger car, large size, petrol, EURO 3 {GLO}| market for | APOS, U	km	0.03km onsite at Sunrise Valley Farm, 64km between SVF and CSC	

Table 3.10 Life cycle carbon-based emissions comparison by feedstock per 10 kg biochar. Emissions of carbon-based greenhouse gasses and mass of carbon emitted per feedstock. For simplicity, emissions under 0.0001 kg were omitted.

Feedstock	Emissions Type	Total Mass Emitted per 10 kg Biochar	Mass of C Emitted per 10 kg Biochar
Softwood	Carbon dioxide, pyrolysis	3.92	1.0584
	Carbon dioxide, fossil	0.697555	0.18834
	Carbon dioxide, biogenic	0.005659	0.001528
	Carbon dioxide, land transformation	0.00038	0.000103
	Carbon monoxide, pyrolysis	2.64	1.13256
	Carbon monoxide, fossil	0.003857	0.001654
	Methane, pyrolysis	0.440057	0.329603
	Methane, fossil	0.000751	0.000563
			Total: 2.71
Hardwood	Carbon dioxide, pyrolysis	6.67	1.8009
	Carbon dioxide, fossil	0.69839	0.188565
	Carbon dioxide, biogenic	0.005666	0.00153
	Carbon dioxide, land transformation	0.000381	0.000103
	Carbon monoxide, pyrolysis	4.49	1.92621
	Carbon monoxide, fossil	0.003861	0.001656
	Methane, pyrolysis	0.750057	0.561793
	Methane, fossil	0.000752	0.000563
			Total: 4.48
Hay	Carbon dioxide, pyrolysis	13.15	3.5505
	Carbon dioxide, fossil	13.33192	3.599619
	Carbon dioxide, biogenic	0.146967	0.039681
	Carbon dioxide, land transformation	0.009056	0.002445
	Carbon monoxide, pyrolysis	8.87	3.80523
	Carbon monoxide, fossil	0.084308	0.036168
	Methane, pyrolysis	1.490729	1.116556
	Methane, fossil	0.015527	0.01163
			Total: 12.16

Table 3.11 100-year global warming potential of life cycle emissions by feedstock per 10 kg biochar. 100-year GWP as determined by the IPCC 100-year GWP impact assessment. For simplicity, emissions under 0.005 kg CO_2eq were omitted.

Feedstock	Emissions Type	GWP (kg CO_2eq)
Softwood	Carbon dioxide, fossil	0.697555
	Carbon dioxide, pyrolysis	3.92
	Dinitrogen monoxide	0.006696
	Methane, fossil	0.022915
	Methane, pyrolysis	13.42174
		Total: 18.1
Hardwood	Carbon dioxide, fossil	0.69839
	Carbon dioxide, pyrolysis	6.67
	Dinitrogen monoxide	0.006704
	Methane, fossil	0.022942
	Methane, pyrolysis	22.87674
		Total: 30.3
Hay	Carbon dioxide, fossil	13.33192
	Carbon dioxide, land transformation	0.009056
	Carbon dioxide, pyrolysis	13.15
	Nitrous oxide	9.489352
	Methane, biogenic	0.010879
	Methane, fossil	0.473565
	Methane, pyrolysis	45.46724
	Methane, tetrafluoro-, CFC-14	0.006986
	Sulfur hexafluoride	0.009007
		Total: 81.95

Table 3.12 Pyrolysis gas yield sensitivity analysis results for 10 kg biochar

Feedstock	Scenario	GWP (kg CO_2eq)	GWP % of Original	Carbon Emitted (kg)	Carbon % Change from Original
Softwood	High gas yield (30%)	26.7	148%	3.97	146%
	Low gas yield (10%)	9.4	51%	1.45	54%
Hardwood	High gas yield (53.3%)	45.2	149%	6.63	148%
	Low gas yield (17.8%)	15.7	51%	2.34	52%
Hay	High gas yield (37.5)	111	135%	16.34	134%
	Low gas yield (12.5%)	52.5	64%	7.92	65%

Table 3.13 Transportation sensitivity analysis results for 10 kg biochar

Feedstock	**Scenario**	**GWP (kg CO_2eq)**	**GWP % of Original**	**Carbon Emitted (kg)**	**Carbon % Change from Original**
Softwood	Low transportation (5km)	17.5	97%	2.56	94%
	Medium transportation (128km)	21	116%	3.49	129%
	High transportation (256km)	24.7	136%	4.45	164%
Hardwood	Low transportation (5km)	29.7	98%	4.33	97%
	Medium transportation (128km)	33.4	110%	5.31	119%
	High transportation (256km)	42.2	139%	7.62	170%
Hay	Low transportation (5km)	73	89%	9.80	81%
	Medium transportation (128km)	82	100%	12.16	100%
	High transportation (256km)	91.3	111%	14.62	120%

Table 3.14 Estimates of biochar carbon residence time in soil over 100 years per 10 kg biochar

Feedstock	Scenario	Mass C Remaining after 100 yr (kg)
Softwood	No C loss (100% retained)	7.29
	Low C loss (90% retained)	6.55
	High C loss (50% retained)	3.64
Hardwood	No C loss (100% retained)	8.42
	Low C loss (90% retained)	7.58
	High C loss (50% retained)	4.21
Hay	No C loss (100% retained)	12.02
	Low C loss (90% retained)	10.82
	High C loss (50% retained)	6.01

Table 3.15 Overall carbon balance (in kg C) over the life cycle of 10 kg biochar for combinations of carbon emissions and soil carbon residence time scenarios. Negative values (in bold) represent a net storage of carbon in soil, while positive values indic ate net loss of carbon over the life cycle.

		Carbon Emissions Scenario				
		Low transportation (5 km)	Medium transportation (128 km)	High transportation (256 km)	Low gas yield (0.5x)	High gas yield (1.5x)
Softwood	No C loss (100% retained)	**-4.72**	**-3.79**	**-2.83**	**-5.83**	**-3.31**
	Low C loss (90% retained)	**-3.99**	**-3.06**	**-2.10**	**-5.10**	**-2.58**
	High C loss (50% retained)	**-1.08**	0.30	0.81	**-2.19**	0.33
Hardwood	No C loss (100% retained)	**-4.09**	**-3.11**	**-0.80**	**-6.08**	**-1.79**
	Low C loss (90% retained)	**-3.25**	**-2.27**	0.04	**-5.24**	**-0.95**
	High C loss (50% retained)	0.12	1.10	3.41	**-1.87**	2.42
Hay	No C loss (100% retained)	**-2.22**	0.14	2.60	**-4.10**	4.32
	Low C loss (90% retained)	**-1.02**	1.34	3.80	**-2.90**	5.52
	High C loss (50% retained)	3.79	6.15	8.61	1.91	10.33

Figures

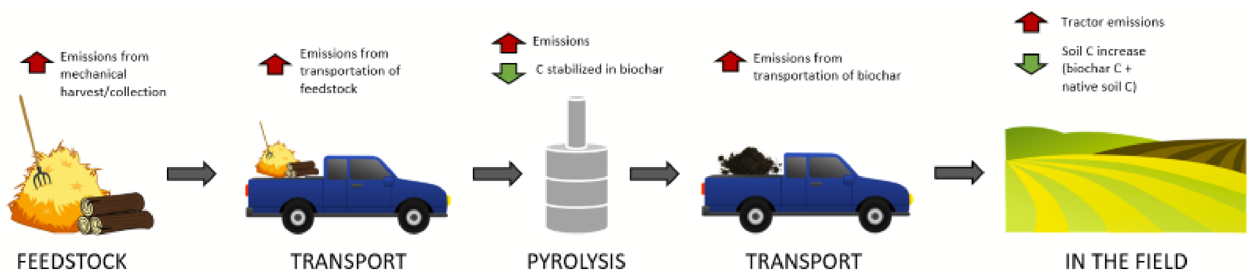

Figure 3.1 Schematic of LCA system boundaries.

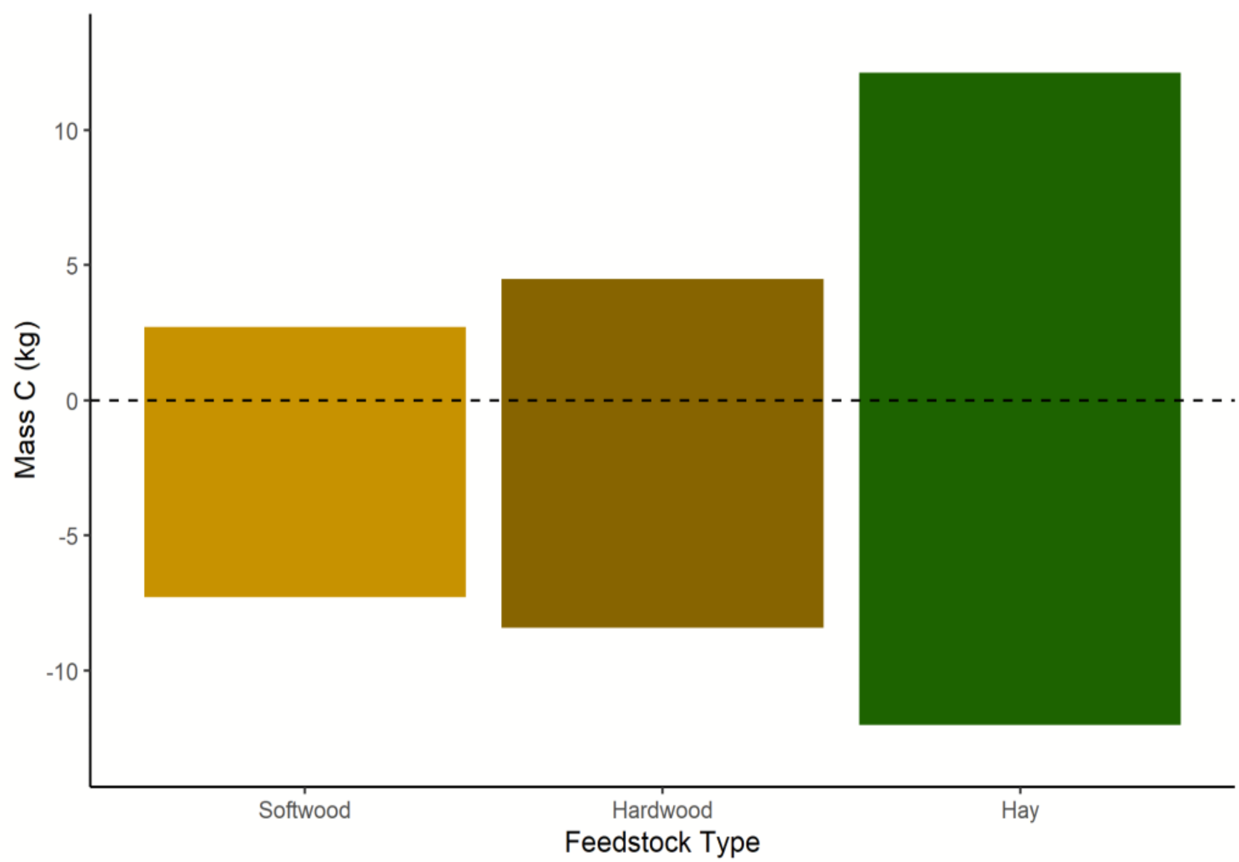

Figure 3.2 Total mass carbon emitted and stored (above control plot carbon levels) in soil over the life cycle of 10 kg biochar. The dotted line represents the divide between emissions (positive values) and additional soil carbon after one year (negative values).

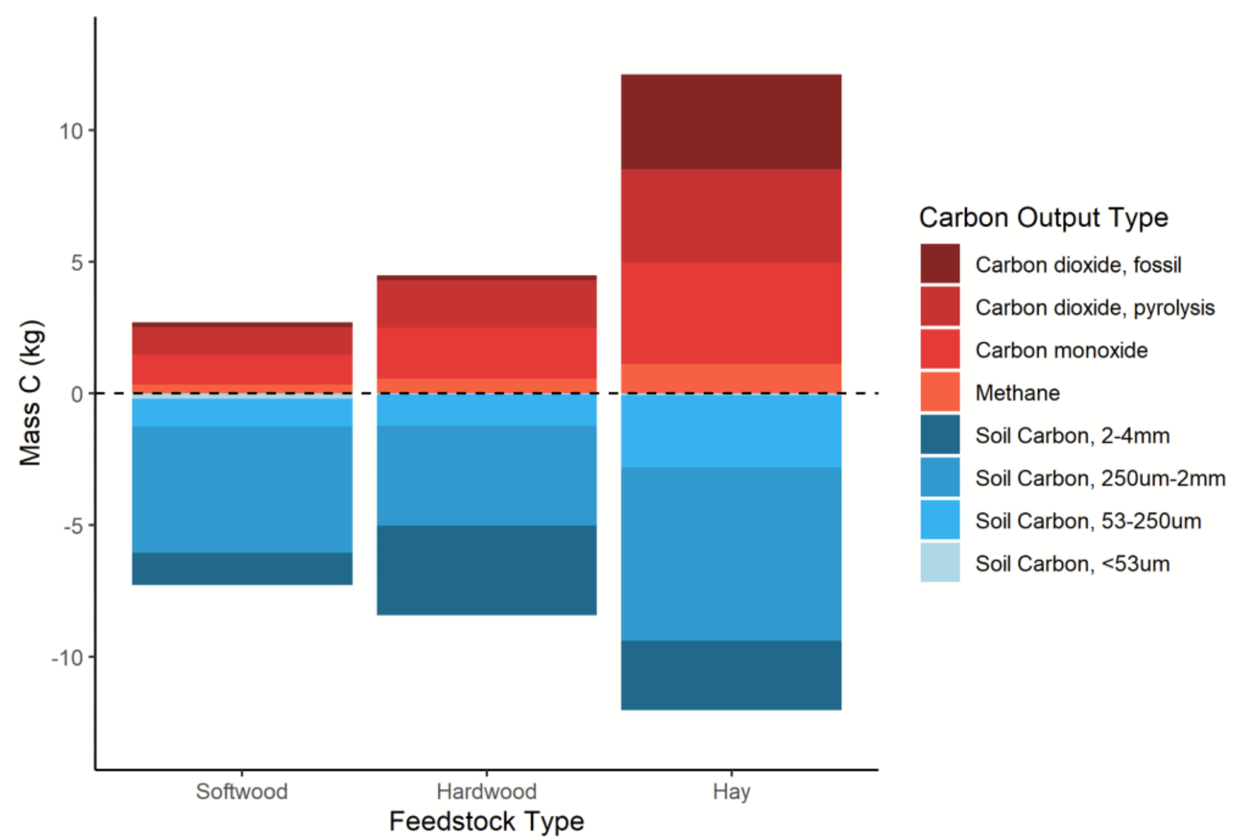

Figure 3.3 Mass carbon emitted and stored in soil (above pre-treatment carbon levels) over the life cycle of 10 kg biochar, by carbon output type. The dotted line represents the divide between emissions (positive values) and additional soil carbon after 1 year (negative values). Note that carbon monoxide and methane sources have been grouped; see Table 3.10 for breakdown of emissions source. Soil fraction carbon values represent increases above pre-treatment levels.

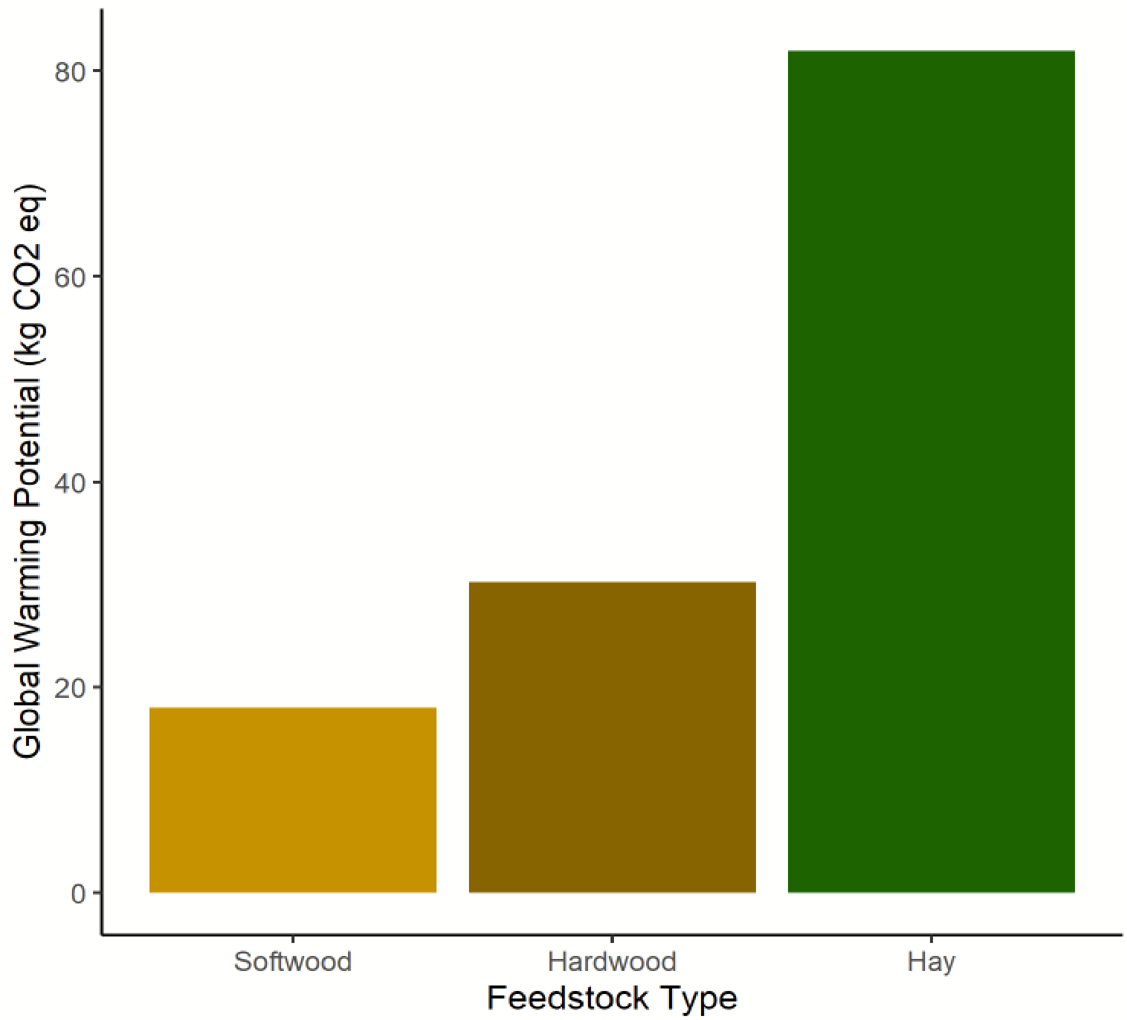

Figure 3.4 100-year global warming potential (kg CO_2eq) of life cycle emissions by feedstock for 10 kg biochar.

CHAPTER 4: Conclusion

The objective of the two studies presented here was to provide useful information on the soil health and carbon storage benefits of biochar amendment on working farms, bridging the gap between scientific research and the growers increasingly interested in using biochar. The first study focused on the impact of biochar on soil health across a range of soil types, climate, and cropping systems in New Mexico, Minnesota, and Virginia. The initial characteristics of both the soil and the biochar applied impacted the outcome. Soil carbon increased by at least 1.35x in all three locations and soil nitrogen content increased in Virginia and Minnesota. Soil pH was relatively unaffected in New Mexico or Virginia, but the acidic, under-pyrolyzed biochar applied in Minnesota drove down soil pH significantly. Electrical conductivity was not impacted at any site.

The second study was a life cycle assessment to quantify and compare the carbon emissions and global warming potential of biochar pyrolyzed from three feedstocks: softwood, hardwood, and hay; applied to soil. These results were paired with soil carbon analysis from the pasture soil plots where the biochar was applied to determine the net carbon balance of each biochar system. The size and sign of each life cycle's carbon balance was determined by the feedstock, the pyrolysis gas yield, and the transportation distances for feedstocks and biochar. Softwood biochar produced the highest net carbon storage potential of the three, followed closely by hardwood biochar. The net carbon storage of these biochars was robust to most of the scenarios tested in a sensitivity analysis. The hay biochar contributed the most to soil carbon storage but also had the highest gross carbon emissions, so it produced a net positive carbon balance across almost all scenarios.

Both studies reinforced the importance of tailoring biochar sourcing and production systems to local conditions, including initial soil properties, feedstock availability, and pyrolysis technology. Ideally, biochar should be made from waste materials up-cycled from local agricultural or silvicultural processes and pyrolyzed in a way that maximizes biochar yield and reduces or repurposes the gas co-products. Feedstocks and pyrolysis methods should be selected with the desired results in mind, whether the priority is improving soil properties or maximizing soil carbon storage. Biochar has the potential to restore soil health and create long-lasting soil carbon pools in agricultural soils, and local-scale biochar systems can and should be at the forefront of the global movement.