Woody biochar’s greenhouse gas mitigation potential across fertilized and unfertilized agricultural soils and soil moisture regimes

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Abstract

Biochar has been widely researched as an important technology for climate smart agriculture, yet work is still necessary to identify the magnitude of potential greenhouse gas (GHG) mitigation and mechanisms involved. This study measured slow-pyrolysis wood-derived biochar’s impact on GHG efflux, mineral N dynamics, and soil organic C in a series of two incubations across fertilized and unfertilized agricultural soils and soil moisture regimes. This research explored the magnitude of biochar’s full GHG mitigation potential and drivers of such impacts. Results of this incubation indicate slow-pyrolysis wood-derived biochar has potential to provide annual emission reductions of 0.58–1.72 Mg CO$_2$-eq ha$^{-1}$ at a 25 Mg ha$^{-1}$ biochar application rate. The greatest GHG mitigation potential was from C sequestration and nitrous oxide (N$_2$O) reduction in mineral N fertilized soils, with minimal impacts on N$_2$O emissions in unfertilized soils, carbon dioxide (CO$_2$) emissions, and methane (CH$_4$) uptake. Analysis of mineral N dynamics in the bulk soil and on biochar isolates indicated that neither biochar impacts on net mineralization and nitrification nor retention of ammonium (NH$_4^+$) on biochar isolates could explain biochar’s N$_2$O reduction. Instead, biochar amendments exhibited consistent N$_2$O emission reductions relative to the N$_2$O emission in the control soil regardless of soil type and fertilization. Results across a soil moisture gradient suggest that woody biochar may aerate soils shifting redox conditions and subsequent N$_2$O production. Understanding the magnitude of biochar’s GHG reduction potential and the mechanisms driving these effects can help inform biochar modeling efforts, explain field results and identify agricultural applications that maximize biochar’s full GHG mitigation potential.

Keywords: ammonium, biochar, greenhouse gas mitigation, methane, nitrate, nitrous oxide, soil respiration

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Introduction

Applying biochar, the solid product of organic matter pyrolysis, as an amendment to agricultural soils can provide an integrated strategy to address waste management, bioenergy production, soil fertility, and GHG mitigation (Lehmann, 2007). Globally, biochar amendments to soils have an estimated potential to provide 1–1.8 Pg CO$_2$-eq in GHG emission reductions (Paustian et al., 2016). In agricultural applications, biochar amendments have potential to reduce GHG emissions by sequestering C in the biochar applied to soils (i.e., due to slower mineralization of the pyrolized biomass to CO$_2$), increasing soil CH$_4$ consumption, decreasing soil N$_2$O emissions, and improving soil fertility to decrease the yield-scaled emissions (Brassard et al., 2016; Mandal et al., 2016; Smith, 2016). Understanding biochar’s impact on individual GHG fluxes across a range of fertilized and unfertilized agricultural soils and soil moisture regimes can help target the most promising applications and identify mechanisms responsible for biochar’s full GHG mitigation potential in soils.

C sequestration is one of the earliest and most widely established benefits of biochar application (Lehmann et al., 2006), stemming from early research on the persistence of black carbon in Terra Preta soils (Glaser et al., 2001). Biochar’s recalcitrant chemical structure (Paris et al., 2005; Spokas, 2010), ability to promote soil aggregation (Soinne et al., 2014), and interaction with mineral phases and soil organic matter (SOM) all contribute to the stabilization of biochar in soils, thus providing a C sink at the centuries timescale (Preston & Schmidt, 2006; Lehmann et al., 2009). Biochar application may also have indirect effects on native soil organic carbon (SOC), either stimulating or preventing mineralization.
of native SOM (Cross & Sohi, 2011; Stewart et al., 2013). A recent meta-analysis of biochar impacts on CO2 in upland soils shows biochar to have a range of responses depending on soil type but overall no significant effect on priming soil C (Liu et al., 2016).

Research on how biochar impacts CH4 production and consumption in agricultural soils is less developed. Generally, agricultural CH4 fluxes are dominated by CH4 emissions from flooded irrigation systems and manure management, while upland agricultural soils provide a moderate CH4 sink (Mosier et al., 1998). A recent meta-analysis of biochar’s effect on CH4 emissions and uptake found that biochar has potential to mitigate CH4 emissions under flood irrigation management regimes but may decrease the CH4 sink in upland agricultural systems (Jeffery et al., 2016). Possible mechanisms influencing CH4 uptake include biochar decreasing the ratio of methanogens to methanotrophs (Feng et al., 2012), possibly through pH mediated affects (Jeffery et al., 2016), directly adsorbing CH4 (Sadasivam & Reddy, 2014) or aerating soils (Karhu et al., 2011).

Agricultural soils are the leading contributor to N2O emissions in many countries world-wide due to microbial transformation of reactive N added through excessive fertilizer application and promotion of N fixing crops (Bouwman, 1996). In laboratory incubations, biochar has been shown to decrease N2O efflux on average by 54%, with contrasting effects by soil type (Cayuela et al., 2014, 2015), but results from field trials are beginning to challenge such findings (Verhoeven et al., 2017). Numerous mechanisms for biochar N2O reductions have been proposed, which can be generally categorized as (1) biochar decreasing nitrification or denitrification rates (van Zwieten et al., 2014; Wang et al., 2015), (2) biochar decreasing the N2O/N2 product ratio of denitrification (Cayuela et al., 2013; Xu et al., 2014; Obia et al., 2015), or (3) biochar impacting diffusion of N2O from soils (Cornelissen et al., 2013; Quin et al., 2015; Harter et al., 2016). While many studies have explored the effects of biochar under aerobic conditions, which are most favorable to N2O production (Baggs & Bateman, 2005), research on biochar’s ability to reduce N2O under aerated soils is more limited. Some studies indicate biochar can actually increase N2O production under aerobic conditions where nitrification may be the dominant N2O formation pathway (Sánchez-García et al., 2014; Wells & Baggs, 2014).

Biochar’s effects on GHG emissions vary greatly depending on biochar feedstock (Mandal et al., 2016). Biochar produced from woody feedstocks has higher porosity and surface area, and lower H:Corg, O:Corg, volatiles and ash contents as compared to other biochars (Enders et al., 2012; Kloss et al., 2012; Ronssse et al., 2013). These characteristics are correlated with increased C stability of biochar (Spokas, 2010; Crombie et al., 2013), increased CH4 oxidation in upland soils (Brassard et al., 2016) and greater N2O emission mitigation potential (Cayuela et al., 2015), making woody biochar a prime candidate for GHG mitigation. In many forested regions of the western United States, there is an oversupply of woody fuels that require management, due to mortality from beetle outbreaks, fire suppression efforts and declining timber markets (Noss et al., 2006; Hicke et al., 2012; Adams, 2013). Producing biochar from these woody fuels and applying it to agricultural soils could serve to produce bioenergy, manage residues, and reduce emissions from agriculture (Field et al., 2013). A mechanistic understanding of how biochar impacts GHG emission across agricultural soils, in interaction with N availability and across soil moisture regimes, is necessary to implement successful biochar GHG mitigation programs. This study explores the GHG mitigation potential of a beetle-killed pine biochar soil amendment along a range of agricultural soils with and without mineral N fertilization and soil moisture regimes and determines how these edaphic factors influence woody biochar’s GHG mitigation potential.

The objectives of this research are to: (1) quantify the full GHG mitigation potential of a woody biochar by identifying the conditions that maximize C sequestration and GHG emission reductions across different agricultural soils with and without fertilization and soil moisture regimes; (2) assess mineral N dynamics within the bulk soil and on biochar isolates to evaluate how soil properties, N fertilization and soil moisture regime impact biochar’s ability to mitigate N2O efflux. We hypothesize that the C sequestration of biochar provides the greatest potential to mitigate GHG emissions when compared on an annualized CO2-eq basis. Relative to C sequestration, we expect biochar to have a significantly smaller impact on CO2 and CH4 emissions. We hypothesize biochar can provide significant reductions in N2O emissions but effects will vary by soil properties, fertilization, and soil moisture. Of the processes impacting soil N2O efflux, we expect biochar to decrease N2O efflux relative to unamended control soils by: (1) reducing net N mineralization, resulting in greater N2O reductions in soils with more total N; (2) causing surface interactions with mineral N in the soil solution to prevent N transformation, resulting in N retention on biochar isolates and greater percent N2O reductions in N fertilized treatments; (3) impacting the relative amounts of N undergoing nitrification vs. denitrification, with biochar treatments exhibiting more aerobic conditions thus decreasing N2O production from denitrification under anaerobic soil moisture regimes.
Materials and methods

We tested these hypotheses in two laboratory incubations where biochar’s full GHG mitigation potential was measured over a 30- or 60-day period. In experiment 1 (E1), under aerobic conditions, we tested how N fertilization and soil properties, including pH, SOM content, and soil texture, regulate biochar’s impacts on GHG efflux and mineral N dynamics. In experiment 2 (E2), on one of the fertilized soils across a range of moisture contents, we tested how soil moisture regime regulates biochar’s impacts on GHG efflux and mineral N dynamics.

Materials

The four agricultural soils were selected to represent different pH, SOM content, and soil texture (Table 1). Soils were sampled to a 10 cm depth using four soil cores per site then samples were consolidated into a single airtight bag. Soil samples were sieved to 2 mm, air-dried, and stored prior to the incubation. Brunauer-Emmett-Teller (BET) surface area analysis was conducted using a single subsample of oven dried soil using a micromeritics BET surface area and porosity analyzer (ASAP 2020; Micromeritics, Norcross, GA, USA). Soil texture was measured on a single 40 g subsample of pre-incubated soils using sedimentation columns (Pansu & Gautheyrou, 2006). Soils were also analyzed for pH, inorganic C, and total C and N as described below.

The woody biochar was produced using a beetle-killed lodgepole pine feedstock, under slow pyrolysis, reaching temperatures of 550°C, by Biochar Now (Berthoud, CO). Biochar was crushed and sieved to obtain a 2–2.8 mm size fraction that could be fractionated and recovered for analysis post-incubation. Biochar BET surface area and pH were analyzed using the same protocol as for the soils. All other biochar analyzes were conducted by Hazen Research, Inc. (Golden, CO, USA) in accordance with International Biochar Initiative protocols (Table 2).

Experimental design

The E1 treatments consisted of the four agricultural soils (CO, ID, ND, and TX; Table 1), two N fertilizer rates (unfertilized and fertilized), and two biochar addition rates (0 and 2.5% by weight) in a fully factorial design with four replicates, for each of the three destructive harvests (day 1, day 30 and day 60; n = 192). An additional destructive harvest was performed on day 7 for all N fertilizer and biochar treatments for the CO and TX soils only (n = 32). A 2.5% biochar application rate was selected to balance trade-offs between economics and GHG effect size (Roberts et al., 2010). This rate is, depending on the soil bulk density (Table 1) and assuming a 10 cm incorporation depth, equivalent to a 23.3–26.5 Mg ha⁻¹ field application.

Samples were prepared using 50 g air-dried soil (0% biochar treatment) or 50 g air-dried mixture of soil and biochar (2.5% biochar treatment) in specimen cups. For E1, all samples were wetted with an amount of distilled (DI) water calculated to achieve 40% water-filled pore space (WFPS) in the unamended control soil, see below, then pre-incubated for 4 days at 25 °C in the dark to activate soil microbial activity. The incubation began on the fifth day by wetting the pre-incubated samples with DI water or a NH₄Cl fertilizer solution to achieve 60% WFPS of the control soil. Fertilized samples received 1 × NH₄Cl at a rate of 140 µg N (g soil)⁻¹ evenly applied to the surface then distributed into the soil samples with the remaining DI water. This rate, depending on soil bulk density, was equivalent to a 130–148 kg ha⁻¹ field N fertilizer application. The sample cups were placed in an airtight, quart-size mason jars, with ports for gas sampling, and stored at 25 °C in the dark throughout the incubation. GHG efflux was sampled throughout the incubation on the treatments scheduled for harvest on day 60, as described below. Samples were destructively harvested at day 1, 7, 30, and 60 and processed for mineral N, as described below. Soil pH, inorganic C, and total C and N analyzes were only performed on the samples harvested at day 60, as described below. Jars were periodically flushed to avoid excessive CO₂ accumulation (>2%), and soil moisture levels were maintained by checking the sample weight to monitor and compensate for evaporative losses.

Table 1 Soil properties for the four temperate agricultural soils used in the incubation, including mean annual temperature (MAT) and mean annual precipitation (MAP) for the site and bulk density (BD), sand (Sa), silt (Si), clay (Cl), total organic carbon (TOC), total inorganic carbon (TIC), total nitrogen (TN) and BET surface area of the soils. Analytical error presented for measurements where available

| Site               | Land Use         | MAT (°C) | MAP (mm) | BD (g cm⁻³) | Sa (%) | Si (%) | Cl (%) | TOC (g kg⁻¹) | TIC (g kg⁻¹) | TN (g kg⁻¹) | pH | BET (m² g⁻¹) |
|--------------------|------------------|----------|----------|-------------|--------|--------|--------|-------------|-------------|-------------|----|-------------|
| Fort Collins, Colorado (CO) | Cultivated maize | 9        | 276      | 1.06 ±0.02  | 35     | 32     | 34     | 0.81 ±0.01  | 0.43 ±0.00  | 0.13 ±0.00  | 7.99 ±0.10 | 36.8         |
| Carey, Idaho (ID) | Rangeland        | 7        | 278      | 0.93 ±0.02  | 28     | 54     | 19     | 5.69 ±0.12  | 0.02 ±0.00  | 0.52 ±0.01  | 5.86 ±0.12 | 11.2         |
| Mandan, North Dakota (ND) | Cultivated wheat | 5        | 402      | 0.94 ±0.00  | 11     | 60     | 29     | 2.38 ±0.01  | 0.03 ±0.00  | 0.24 ±0.00  | 7.27 ±0.06 | 20.0         |
| Vernon, Texas (TX) | Cultivated wheat | 17       | 665      | 1.00 ±0.01  | 14     | 50     | 36     | 0.91 ±0.01  | 0.03 ±0.01  | 0.11 ±0.00  | 8.04 ±0.26 | 49.4         |

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Table 2  Biochar production information along with physical and chemical properties

| Description         | Biochar property |
|---------------------|------------------|
| Feedstock           | Beetle-killed lodgepole pine |
| Pyrolysis           | Slow pyrolysis, 550 °C max temp |
| Particle Size       | Sieved to 2.0–2.8 mm |
| Organic C           | 86.2 g/100 g biochar |
| H:Corg              | 0.35 molar ratio |
| C:N                 | 478.8 mass ratio |
| Ash                 | 1.1 g/100 g biochar |
| pH                  | 8.49 |
| BET                 | 111.89 m² g⁻¹ biochar |

The soil moisture gradient incubation (E2) was conducted using the N fertilized (1 m NH₄Cl at a rate of 140 µg N (g soil)⁻¹) CO soil with treatments consisting of two biochar rates (0 and 2.5% by weight) and four soil moisture levels (40%, 60%, 80%, and 100% WFPS of the control soils, see below) in a fully factorial design with four replicates and one harvest at day 30. Samples were incubated using the same protocol as E1 and wetted to the desired WFPS after the pre-incubation period (note that the dry treatments were pre-incubated to 30% WFPS of the control). Samples were destructively harvested at day 30, and analyzed for mineral N as described below.

Porosity of treatments was calculated assuming a soil particle density of 2.65 g cm⁻³ and biochar solid C density of 1.6 g cm⁻³ (Downie et al., 2009). In accordance with the methods used by Quin et al. (2015), the porosity of the soil and biochar mixtures was calculated as the composite from the estimated porosities of the components. Soil moisture was added to achieve 40%, 60%, 80%, or 100% WFPS of the control soils, resulting in an estimated WFPS for the biochar treatments of 39.6%, 59.2%, 79.1%, and 98.7%, respectively. These treatments are referred to as dry (39.6–40% WFPS), aerobic (59.2–60% WFPS), moderately anaerobic (79.1–80% WFPS), and anaerobic (98.7–100% WFPS) throughout the text.

**GHG Efflux**

The net CO₂, CH₄, and N₂O fluxes were measured in both E1 and E2 by sampling the headspace gas periodically throughout the incubation (for E1 on day 1, 2, 3, 4, 6, 8, 10, 13, 16, 20, 25, 30, 35, 42, 50, 58; for E2 on day 0.5, 1, 1.5, 3, 4, 5, 6, 8, 10, 12.5, 15, 17, 20, 24, 28). Headspace GHG concentrations were sampled using cavity ring down spectroscopy (CRDS) with a recirculation pump (G2508 CRDS Analyzer; Picarro Inc., Santa Clara, CA, USA). GHG concentrations were adjusted to account for mixing of headspace gas with the gas in the analyzer lines and chamber when sample ports were connected in a closed loop. The equilibrated GHG concentration of ambient air in the lines and chamber was measured prior to mixing with sample headspace. Mixed GHG samples were left to equilibrate for 3 min before taking measurements. Headspace GHG concentrations were then calculated using a mixing model (Eqn 1).

$$[\text{GHG}]_h = \frac{n_m[\text{GHG}]_{im} - n_c[\text{GHG}]_{lc}}{n_h},$$ (1)

where [GHG]ₜₘ is the GHG concentration of the headspace, [GHG]ₜₘ is the GHG concentration of the lines and chamber, and [GHG]ₜₘ is the GHG concentration of the mixed sample, nₘ is the mol gas in headspace, nₙ is the mol gas in lines, and chamber and nₘ are the sum of nₘ and nₙ. GHG efflux was calculated as the difference between GHG concentration of the headspace after flushing, and the GHG concentration at the next sampling date.

**Soil and biochar analysis**

Upon harvest, each sample was thoroughly mixed before subsampling for analysis of extractable mineral N in bulk soil (10 g), extractable mineral N in biochar isolates (10 g), pH (5 g), and inorganic C, total C and N concentrations (5 g). Soil subsamples for mineral N analyzes were processed prior to air drying. Soil subsamples for inorganic C, total C and N analyzes were ground using mortar and pestle and oven dried at 63 °C prior to analysis.

Soil pH was measured in DI water (soil : water ratio, 5 : 1 by mass) using a pH electrode and stirrer bar (Orion EA940 Expandable ionAnalyzer; Orion Research, Jacksonville, FL, USA). Inorganic C was only measured on the E1 CO soils (the other soil types had <0.1% inorganic C; Table 1) by dissolving samples in 6 N HCl +3% ferrous chloride and sampling CO₂ produced using a pressure transducer (Setra 280E; Boxborough, MA, USA; Sherrod et al., 2002). Total C and N were measured using an elemental analyzer (LECO Tru-SPEC elemental analyzer; Leco Corp., St. Joseph, MI, USA). All analyzes for the soils prior to incubation were performed on three laboratory replicates, while analyzes of the E1 day 60 soils were performed on each of the four replicates.

To quantify extractable mineral N concentrations in the bulk soil, a 10 g subsample of the bulk soil (E1 day 1, 7, 30, and 60; E2 day 30) was extracted using 2 m KCl (soil : KCl ratio, 5 : 1 by mass), shaken for 1 h, filtered (Whatman #40 ashless filter paper), and analyzed colorimetrically (Alpkem Flow Solution IV Automated wet chemistry system; O.I. Analytical, College Station, TX, USA) for NH₄⁺ and nitrate (NO₃⁻).

To quantify the extractable mineral N within the biochar particles, on the 2.5% biochar treatments, another 10 g subsample was taken (E1 day 1, 7, 30, and 60; E2 day 30) and placed on a 2-mm sieve and rinsed with 1 L DI water to isolate the biochar from the bulk soil (as biochar particle size was 2–2.8 mm). The biochar isolates were then extracted and analyzed for mineral N using the same method as described above. The dry mass of biochar isolates in the extracted subsample was determined by recovering the biochar post extraction, drying overnight at 63 °C and weighing, accounting for any KCl residues on the biochar.

**Data and statistical analysis**

Biochar-induced GHG emission reductions were calculated as the difference between biochar and control GHG emissions,
then converted to CO₂ equivalent using 100-year global warming potentials (Myhre et al., 2013). The standard error for the GHG emission reductions was calculated using error propagation of the standard error for each treatment. An annualized estimate for C sequestration from the biochar addition was estimated by multiplying the biochar C added by a factor to estimate the percent biochar C left after 100 years, determined from a regression of the H:Corg of the biochar by Zimmerman and Gao (2013) \(-74.3 \times \text{(H:Corg)} + 110.2 = 84.2\%\), then dividing over 100 years to annualize.

Treatment effects were analyzed on the following response variables; cumulative CO₂, CH₄, N₂O efflux, SOC, total N, pH, bulk soil NH₄⁺ and NO₃⁻ and biochar isolate NH₄⁺ and NO₃⁻. We used a three factor ANOVA for E1 with fertilizer, amendment and soil as factors, and a two factor ANOVA for E2 with amendment and soil moisture level as factors. We used an ANCOVA to test for significant differences between soil and fertilization treatments on the relationship between N₂O emissions in the biochar vs. control soils, by modeling N₂O in the biochar treatments using soil and fertilizer as factors and N₂O in the control treatments as a covariate. Linear and exponential regressions were applied to model cumulative CO₂ emissions by initial SOC content, cumulative N₂O emissions by initial TN content, and biochar isolate NO₃⁻ concentrations by bulk soil NO₃⁻ concentrations. For mineral N concentrations, NH₄⁺ concentrations with mean values less than the measurement uncertainty were reported as undetectable. Where response variables were non-parametric, a Box Cox parametric power transformation was applied to transform the data for analysis (Box & Cox, 1964). All statistical analyzes were performed in R version 3.3.1 (R Core Team, 2016). Significant differences between treatments were determined where \( P < 0.05 \). All error bars are reported as the standard error.

**Results**

**Biochar’s impact on C sequestration and GHG efflux by soil types and fertilization**

All samples continued to respire throughout the 60-day incubation with CO₂ emissions never reaching an asymptote (Fig. 1a, b). Soil type significantly affected cumulative CO₂ emissions \(( P < 0.001 )\) with a strong positive correlation between initial SOC content and cumulative CO₂ emissions \( ( R^2 = 0.83; P < 0.001 )\). However, on a percent initial SOC basis, the TX soil experienced the greatest C loss as CO₂ (5.3%), followed by CO (3.1%), ID (2.6%), and ND (2.1%) soils. N fertilization significantly decreased soil respiration by 18.3% \(( P < 0.001 )\). Overall, biochar amendments did not have a significant impact on soil CO₂ emissions, but there were significant interactions with soil type and fertilization (Table 3). Biochar resulted in a 3.6% decrease in CO₂ in the TX soils \(( P = 0.03 )\) and a 0.6% decrease in CO₂ emissions. Applied to transform the data for analysis (Box & Cox, 1964). All statistical analyzes were performed in R version 3.3.1 (R Core Team, 2016). Significant differences between treatments were determined where \( P < 0.05 \). All error bars are reported as the standard error.

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![Fig. 1](image-url) Cumulative GHG efflux dynamics from biochar (closed shapes) and control (open shapes) treatments for the CO (orange squares), TX (red circles), ND (blue triangles), and ID (green diamonds) soils by nitrogen fertilization (unfertilized, a, c, e; fertilized, b, d, f) and GHG type (CO₂, a, b; CH₄, c, d; N₂O, e, f) over the 60-day incubation (E1). Error bars display standard error.

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the fertilized treatments ($P = 0.04$) but the other soils and unfertilized treatments showed no significant biochar effects on respiration (Fig. 1a, b). The cumulative CO$_2$-C losses from this short 60-day incubation constitute 5.9%, 2.2%, 2.0%, and 1.3% of the initial biochar C added for the ID, ND, TX, and CO soils respectively. Biochar C remaining after 100 years was estimated, using the H:C org molar ratio regression from Zimmerman and Gao (2013), at a total of 907.3 mg C per sample.

All treatments experienced net CH$_4$ uptake in E1 but with relatively low cumulative uptake ($-2.80 \pm 0.37$ to $-0.08 \pm 0.07$ μg CH$_4$ treatment$^{-1}$), especially when reported on a CO$_2$-eq basis ($-95$ μg CO$_2$-eq uptake per treatment). Biochar overall increased CH$_4$ uptake by 48.6% ($P < 0.001$), with significant interactions with soil type and fertilization (Table 3). Across soils, biochar increased CH$_4$ uptake by 154.3% in the ID soil ($P = 0.01$), 60.0% in the ND soil ($P < 0.001$) and 43.3% in the TX soil ($P < 0.001$) but had no impact on the CO soil. Between N fertilization treatments, biochar showed greater potential to increase CH$_4$ uptake by 57.4% in the unfertilized treatments ($P < 0.001$) compared to 38.3% increase in uptake under the fertilized treatments ($P = 0.46$) or fertilizer ($P = 0.16$) on this relationship (Fig. 2).

**Biochar’s impact on mineral N dynamics by soil types and fertilization**

NH$_4^+$ concentrations in the unfertilized soils were low throughout the incubation with NH$_4^+$ only detected in N$_2$O efflux and initial total N ($R^2 = 0.82$; $P < 0.001$). Biochar amendments decreased N$_2$O efflux in the fertilized ID, ND, TX, and CO soils by 63.6% ($P = 0.04$), 56.5% ($P = 0.03$), 48.9% ($P = 0.26$), and 10.8% ($P = 0.63$), respectively. In the unfertilized soils, biochar amendments led to a decrease of 59.8% ($P < 0.001$), 57.0% ($P < 0.001$), 52.9% ($P = 0.08$), and 8.5% ($P = 0.56$) in the ND, TX, ID, and CO soils, respectively. N$_2$O efflux in the biochar treatments exhibited a linear relationship with N$_2$O efflux in the control treatment ($R^2 = 0.81$; $P < 0.001$), and there was no significant effect of soil ($P = 0.46$) or fertilizer ($P = 0.16$) on this relationship (Fig. 2).

**Table 3** Model of the response of greenhouse gas emissions (CO$_2$, CH$_4$ and N$_2$O) to biochar (B), fertilization (F) and soil type (S) treatments in Experiment 1 (see text for details). Degree of freedom (df) $F$ and $P$-values from a three factor ANOVA are reported, with significant values ($P < 0.05$) bolded.

| Effect | CO$_2$ | CH$_4$ | N$_2$O |
|--------|--------|--------|--------|
| B      | df     | $F$    | $P$    | $F$    | $P$    | $F$    | $P$    |
| B      | 1      | 0.92   | 0.34   | 70.75  | <0.01  | 37.22  | <0.01  |
| F      | 1      | 360.53 | <0.01  | 120.36 | <0.01  | 1.16E3 | <0.01  |
| S      | 3      | 6.63E3 | <0.01  | 405.90 | <0.01  | 141.77 | <0.01  |
| B x F  | 1      | 3.88   | 0.05   | 5.61   | 0.02   | 3.92   | 0.05   |
| B x S  | 3      | 2.83   | 0.05   | 50.55  | <0.01  | 3.04   | 0.04   |
| F x S  | 3      | 109.20 | <0.01  | 125.99 | <0.01  | 118.41 | <0.01  |
| B x F x S | 3 | 1.54   | 0.22   | 0.61   | 0.62   | 1.82   | 0.16   |

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the bulk soil on day 1 ID and ND soils, then decreasing to 0 by day 60 (Fig. 3a). NO$_3^-$ concentrations in the bulk soil of the unfertilized treatments gradually rose throughout the incubation with a net increase of 10.3 ± 0.6, 16.8 ± 0.6, 20.0 ± 0.4, and 48.7 ± 0.9 μg NO$_3^-$-N (g soil)$^{-1}$, in the TX, ND, CO, and ID control soils, respectively (Fig. 3c). In the fertilized treatments, where 140 μg NH$_4^+$-N (g soil)$^{-1}$ was added, the day 1 soils captured this NH$_4^+$ addition in the bulk soil, then concentrations decreased to 0 by the end of the incubation in all soils except for the ID soil (Fig. 3b). Over the 60-day incubation bulk soil NO$_3^-$ concentrations increased even more than the unfertilized treatments (Fig. 3c, d), with a net increase of 39.8 ± 1.7, 48.4 ± 1.4, 50.8 ± 1.0, and 74.5 ± 1.4 μg NO$_3^-$-N (g soil)$^{-1}$, in the TX, ND, CO, and ID control soils, respectively. The additional accumulation of NO$_3^-$-N in the fertilized treatments relative to the unfertilized treatments can account for 18.4–24.4% of the NH$_4^+$-N added, indicating the NH$_4$Cl fertilizer underwent other transformation pathways.

Biochar treatments exhibited similar dynamics as control treatments where NH$_4^+$ decreased to 0 for all but the ID soils. Across soil types, in both the fertilized and unfertilized treatments at day 60, biochar reduced bulk soil NO$_3^-$ relative to the control by 3.0% ($P = 0.01$) and 11.6% ($P < 0.001$), respectively. However, such differences in bulk soil NO$_3^-$ were not significant across soil types at day 1 or day 30. Only the CO soils showed consistent reductions in bulk soil NO$_3^-$ concentrations relative to the control soils throughout the incubation, with some of the unfertilized soils also indicating biochar reduced NO$_3^-$ at day 60 in the unfertilized treatments (Fig. 3c; Table 4).

Mineral N dynamics on the biochar isolates displayed similar trends as mineral N in the bulk soils (Fig. 4). In the unfertilized soils, NH$_4^+$ concentrations on biochar isolates were below the detection limit for all treatments. In the fertilized treatments, NH$_4^+$ recovery on biochar isolates could account for 0.8–1.1% of the total NH$_4^+$ measured in the day 1 fertilized treatments. However, by day 60 in the fertilized there was little evidence for NH$_4^+$ retention on biochar isolates with NH$_4^+$ concentrations dropping to <4.49 ± 2.57 μg NH$_4$-N (g biochar)$^{-1}$. Given the small biochar addition rate such NH$_4^+$ would not have been detectable in the bulk soil extracts.

NO$_3^-$ concentrations on biochar isolates were significantly higher than bulk soil on a per mass basis; therefore, NO$_3^-$ concentrations on biochar isolates were compared on a per surface area basis. NO$_3^-$ concentrations by bulk soil could predict NO$_3^-$ concentrations on biochar isolates following a linear relationship ($R^2 = 0.84; P < 0.001$). NO$_3^-$ recovery on biochar isolates accounted for 7.0%, 8.5%, 9.3%, and 12.4% of the total NO$_3^-$ extracted from bulk soils across the CO, ND, TX, and ID soils, respectively. Compared to biochar’s percent surface area for each soil type, 7.2%, 12.5%, 5.5%, and 20.4% of the total surface area in the CO, ND, TX and ID soils, respectively (calculated using BET surface areas; Table 1), biochar isolates retained less NO$_3^-$ per surface area in the higher SOM ND and ID soils, and similar to more NO$_3^-$ per surface area in the CO and TX soils.

Fig. 3 Soil mineral nitrogen (N) extracted from the bulk soil for biochar (closed shapes) and control (open shapes) treatments for the CO (squares), TX (circles), ND (triangles), and ID (diamond) soils by N fertilization (unfertilized, a, c; fertilized, b, d) and mineral N type (ammonium (NH$_4^+$), a, b; nitrate (NO$_3^-$), c, d) measured over the course of the 60-day incubation (E1). Error bars display standard error.

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Soil total N was not significantly impacted by biochar amendments but fertilization significantly increased total N in the TX, ND, and CO soils (Table S1). Biochar and fertilization significantly altered pH with interactions by soil type (Table S1), where biochar increased pH by 1.1% \((P < 0.001)\) and fertilization decreased pH by 6.5% \((P < 0.001)\) across treatments. In the unfertilized ID and ND soils, which had a lower initial pH (Table 1), biochar significantly increased pH by 1.5% \((P = 0.004)\) and 2.9% \((P = 0.003)\), respectively. In the ID soil, which was the only acidic soil used in this study, biochar also increased pH in the fertilized treatment by 1.9% \((P < 0.001)\).

Biochar’s impact on GHG efflux by soil moisture regime

In E2, the soil moisture and biochar treatments, as well as their interaction, significantly affected GHG efflux, with some exceptions for CH\(_4\) (Fig. 5; Table 5). In the control soils along the soil moisture gradient, soil respiration peaked in the aerobic treatment then declined in the anaerobic treatments. In the biochar soils, soil respiration continued to increase up to the moderately anaerobic treatment and only slightly decreased in the anaerobic treatment. Such interactions resulted in biochar significantly increasing soil respiration by 11.1% \((P = 0.01)\) and 46.1% \((P < 0.001)\). CH\(_4\) dynamics in this incubation resulted in small \((0.04 \pm 0.02 \text{ to } 0.09 \pm 0.004 \mu g \text{ CH}_4)\) but net CH\(_4\) emissions from soil. Biochar amendments led to a 138.6% increase in CH\(_4\) efflux \((P = 0.003)\) in the anaerobic treatment but had no impact on CH\(_4\) at any of the other soil moisture levels. Surprisingly, water content did not significantly affect CH\(_4\) fluxes, which however suffered in this experiment from high within treatment variability (Fig. 5c, d; Table 5). The soil moisture gradient also highlighted the large impact soil moisture, biochar, and their interactions have on N\(_2\)O (Fig. 5e, f; Table 5). In the control soils, the anaerobic treatments (Fig. 5f) resulted in significantly greater N\(_2\)O emissions than the dry and aerobic treatments (Fig. 5e). In the moderately anaerobic treatment, biochar amendments resulted in the largest reduction in N\(_2\)O emissions relative to the

| Soil | Unfertilized | Fertilized |
|------|--------------|------------|
|      | Day 1        | Day 30     | Day 60     | Day 1        | Day 30     | Day 60     |
| CO   | \(-1.19 (P = 0.01)\) | \(-1.32 (P = 0.04)\) | \(-2.49 (P = 0.01)\) | \(-1.09 (P = 0.04)\) | \(-1.01 (P = 0.47)\) | \(-3.74 (P = 0.02)\) |
| TX   | \(0.06 (P = 0.81)\) | \(1.47 (P = 0.23)\) | \(-1.09 (P = 0.06)\) | \(0.34 (P = 0.25)\) | \(3.57 (P < 0.01)\) | \(-1.37 (P = 0.30)\) |
| ND   | \(-0.15 (P = 0.80)\) | \(1.84 (P = 0.65)\) | \(-3.86 (P < 0.01)\) | \(-0.18 (P = 0.78)\) | \(2.57 (P = 0.10)\) | \(-1.59 (P = 0.34)\) |
| ID   | \(0.44 (P = 0.28)\) | \(-1.30 (P = 0.25)\) | \(-8.99 (P < 0.01)\) | \(-0.19 (P = 0.64)\) | \(-2.04 (P = 0.25)\) | \(-1.30 (P = 0.51)\) |

Fig. 4 Soil mineral nitrogen (N) extracted from biochar isolates in biochar treatments for the CO (squares), TX (circles), ND (triangles), and ID (diamonds) soils by N fertilization (unfertilized, a, c; fertilized, b, d) and mineral N type (ammonium (NH\(_4^+\)), a, b; nitrate (NO\(_3^-\)), c, d) measured over the course of the 60-day incubation (E1). Error bars display standard error.

Table 4 Effect size and \(P\)-values for the difference between NO\(_3^-\) extracted from the bulk soil in the biochar treatments relative to the control treatments by fertilization treatments, soil types and day sampled, with significant values \((P < 0.05)\) bolded.
control both in magnitude and percent change (649.40/88.63 l gN2O, 86.9% decrease; \( P < 0.01 \)). In the anaerobic treatment, N2O efflux from the biochar amended soils was not significantly different from the control soils (\( P = 0.74 \)).

**Biochar’s impact on mineral N by soil moisture regime**

At the end of the 30-day soil moisture incubation, bulk soil NH4+ concentrations were below detection limits in all treatments (0 to 0.93 ± 0.33 μg NH4⁺-N (g soil)⁻¹) except for the anaerobic control treatment where 27.87 ± 0.51 μg NH4⁺-N (g soil)⁻¹ remained (Fig. 6a). In the control treatments, bulk soil NO3⁻ concentrations initially increased along the soil moisture gradient up to the aerobic treatment but then began to decrease under the more anaerobic soil moisture regimes. Biochar bulk soil NO3⁻ dynamics differed from the control soil across the soil moisture gradient, where concentrations steadily increased as soils became more anaerobic (Fig. 6b). This resulted in biochar increasing NO3⁻ by 13.1% (\( P = 0.002 \)) in the moderately anaerobic treatment and 60.7% (\( P < 0.001 \)) in the anaerobic treatment. Mineral N dynamics on the biochar isolates in E2 followed similar patterns as in E1. NH4⁺ concentrations on biochar isolates were below detection limit for all moisture levels (detection limit 5.4 μg NH4⁺ (g biochar)⁻¹). NO3⁻ concentrations on biochar isolates exhibited slightly different dynamics from bulk soil where NO3⁻ increased from the dry to moderately anaerobic treatment but decreased slightly in the anaerobic treatment (Fig. S1).

**Biochar’s GHG mitigation potential**

Overall, in our study, woody biochar’s annualized C sequestration provided the greatest average GHG...
benefit of 33.2 mg CO$_2$-eq per 50 g soil + biochar treatment followed closely by biochar’s N$_2$O reductions in the fertilized treatments of 22.9 ± 12.8 mg CO$_2$-eq per treatment. Biochar’s ability to reduce CO$_2$ emissions across treatments and N$_2$O in the unfertilized treatments was both negligible at 1.2 ± 2.4 and 2.2 ± 3.0 mg CO$_2$-eq per treatment, respectively, with the high variance highlighting uncertainty in biochar’s ability to mitigate GHG efflux in unfertilized soils. Biochar did significantly improve CH$_4$ uptake, but the GHG reduction was orders of magnitude lower at 10.2 ± 6.2 µg CO$_2$-eq per treatment. Biochar’s full GHG mitigation potential showed marked differences by soil type and fertilization especially for N$_2$O (Fig. 7), which emphasizes the importance of understanding controls on biochar’s ability to mitigate N$_2$O.

Depending on fertilization and soil environmental conditions (assuming predominately aerobic conditions), scaling the results from this incubation indicate that biochar amendments showed potential to provide an annual GHG benefit from 0.58 to 1.72 Mg CO$_2$-eq ha$^{-1}$ at a 25 Mg ha$^{-1}$ biochar application rate or 23.2 to 68.7 kg CO$_2$-eq ha$^{-1}$ (Mg biochar)$^{-1}$. GHG mitigation potential from E2 highlighted biochar’s significant GHG mitigation potential for N$_2$O under moderately anaerobic conditions (Fig. 8). By contrast, under anaerobic conditions biochar induced an increase in GHG emissions from increased CO$_2$ efflux.

**Discussion**

To maximize biochar’s GHG mitigation potential from bioenergy to agricultural systems, it is important to understand biochar’s effect on individual GHG fluxes and C sequestration, then target biochar applications to achieve the greatest net GHG emission reduction. Similar to other studies measuring woody biochar impacts on different GHG emission reductions, this study...
showed the greatest GHG mitigation potential in C sequestration and N\textsubscript{2}O emission reductions (Zheng et al., 2012; Stewart et al., 2013; Thomazini et al., 2015). These incubations highlight the magnitude of biochar’s effect on mitigating different GHGs, rather than simply the percent change. Across different agricultural soils, biochar application for GHG mitigation shows the most promise for C sequestration and reducing N\textsubscript{2}O emissions in fertilized soils, while impacts on CO\textsubscript{2} priming and CH\textsubscript{4} prove to be minimal.

**CO\textsubscript{2} impacts and C sequestration**

Biochar’s ability to positively or negatively prime native SOM is one concern researchers have identified when considering biochar’s full GHG mitigation potential (Liu et al., 2016). Biochar has been predicted to stimulate decomposition of native SOC due to inputs of labile biochar C or microbial protection from predators stimulating mineralization (Cross & Sohi, 2011). Negative priming could also occur through direct sorption of labile organic matter on biochar or within biochar-mineral fractions (Lu et al., 2014; Singh & Cowie, 2014). While this study did not allow for a differentiation of biochar-derived from soil-derived CO\textsubscript{2}-C and tested a limited number of soils in a short-term incubation, it found, in accordance with the literature (Liu et al., 2016), that there was no evidence of biochar significantly increasing CO\textsubscript{2} emissions. Two of the soils with lower SOC exhibited biochar decreasing CO\textsubscript{2} efflux, suggesting biochar functional groups may interact with labile SOM to decrease mineralization (Pignatello et al., 2006; Jiang et al., 2016). However, on a CO\textsubscript{2}-eq basis these biochar-induced emissions are minimal comprising <0.13% of the total estimated C sequestration (or 13.4% of the annualized estimated C sequestration). Methodologies to quantify biochar’s GHG mitigation potential should address the potential effects of priming, but estimations from the literature (Liu et al., 2016) can be used to streamline monitoring requirements.

**CH\textsubscript{4} impacts**

This woody biochar’s 48.6% increase in the CH\textsubscript{4} sink contrasts findings of the recent Jeffery et al. (2016) biochar CH\textsubscript{4} meta-analysis which found biochar decreases the CH\textsubscript{4} sink in upland soils. In this aerobic incubation, biochar showed greater potential to increase CH\textsubscript{4} uptake in the unfertilized treatment relative to the fertilized soils, which may indicate NH\textsubscript{4}\textsuperscript{+} is competitively excluding CH\textsubscript{4} at binding sites preventing CH\textsubscript{4} oxidation (Bedard & Knowles, 1989). However, biochar’s CH\textsubscript{4} mitigation potential for upland agricultural soils must be viewed in the context of the magnitude of change in addition to percent change. While biochar did significantly increase the CH\textsubscript{4} sink, this net GHG benefit was still <0.04% of the annualized GHG benefit from C sequestration. Biochar application to upland agricultural systems therefore provides limited net GHG mitigation from increasing CH\textsubscript{4} uptake relative to other GHGs.

Along the soil moisture gradient in the fertilized CO\textsubscript{2} soils (E2) biochar led to an increase in soil CH\textsubscript{4} efflux ($P = 0.03$), again contrasting the trend of decreased CH\textsubscript{4} in flooded systems (Jeffery et al., 2016). Increases in CH\textsubscript{4} have also been found in numerous other studies citing mechanisms such as inputs of labile C from biochar under anaerobic conditions stimulating CH\textsubscript{4} production (Zhang et al., 2013) and reduced CH\textsubscript{4} oxidation from changes in microbial community composition (Spokas & Reicosky, 2009; Yu et al., 2013). However, further exploration of different biochars in saturated soils may provide larger magnitudes of CH\textsubscript{4} emission reduction that could serve as a viable GHG mitigation strategy (Feng et al., 2012; Jeffery et al., 2016).

**N\textsubscript{2}O impacts**

Relative to impacts on other GHG emissions, this woody biochar shows the greatest GHG mitigation potential in reducing N\textsubscript{2}O emissions from fertilized soils. The interactions between biochar and SOM, soil pH, soil texture, fertilization, and soil moisture on mineral N dynamics and N\textsubscript{2}O efflux can help inform the mechanisms by which biochar impacts N\textsubscript{2}O efflux. Hypotheses for biochar’s ability to reduce N\textsubscript{2}O include...
effects on nitrification and denitrification rates (Wang et al., 2015; He et al., 2016), sorption of mineral N to the biochar (Sohi et al., 2010; Case et al., 2012; van Zwieten et al., 2014) and impacts on soil aeration (Rogovskaya et al., 2011; Augustenborg et al., 2012; Mukherjee et al., 2014). These incubations explored such hypotheses by looking at impacts on mineral N dynamics and the resulting N₂O efflux across treatments.

We found that:

1 Woody biochar did not reduce N₂O emissions by reducing mineral N availability

Biochar has been hypothesized to decrease N₂O by reducing net N mineralization and thus the N available for N₂O production (Wang et al., 2015; He et al., 2016). We evaluated this impact in the unfertilized treatments by comparing biochar’s impact on net N mineralization and nitrification, and N₂O efflux. For the unfertilized soils, the positive correlation between soil total N and N₂O production and accumulation of NO₃ throughout the incubation suggests that N mineralization and subsequent nitrification may be an important control on N₂O production. During the first 30 days of the incubation biochar significantly decreased N₂O efflux in the ID and TX soils, yet NO₃ availability in these soils was not impacted (Fig. 3c; Table 4). Only the CO soil exhibited reduced NO₃ availability by day 30 but this soil showed no significant reduction of N₂O. The unfertilized treatments did experience a significant decrease in NO₃ by day 60 (Fig. 3c; Table 4), but 67% to 100% of the N₂O was produced by day 30 with a clear reduction in N₂O across biochar treatments during this period (Fig. 1e, f). The unfertilized treatments thus provide little support for the claim that biochar’s N₂O reduction is driven by biochar reducing mineral N availability through decreased net N mineralization.

Reducing available N through N sorption processes on biochar surfaces is another commonly cited mechanism for biochar’s ability to reduce soil N₂O efflux (Case et al., 2012; Sohi et al., 2010; Thomazini et al., 2015; van Zwieten et al., 2010). Leachate studies have demonstrated that biochar surfaces can retain mineral N, potentially through surface reactions that prevent microbial transformations (Cui et al., 2016). In our study, NH₄⁺ recovery from both the bulk soil and biochar isolates was minimal after day 1, with no retention of extractable NH₄⁺ on biochar isolates. Extractable NO₃⁻ recovered on biochar isolates also exhibited a strong correlation relative to the NO₃⁻ concentrations of the bulk soil. This indicates that sorption processes impacting nitrification are not required to explain extractable N recovered on the biochar surfaces, and mineral N dynamics within the bulk soil can account for mineral N dynamics on biochar isolates. Other studies confirm that biochar does have significant potential to sorb NH₄⁺, but sorption is primarily mediated through cation exchange processes (Zeng et al., 2013; Cui et al., 2016), where mineral N would still be accessible to microbial transformation. Physical adsorption within interior biochar pores has also been shown to be an important mechanism for the adsorption capability of biochar (Zhang et al., 2012; Nguyen et al., 2017). Biochar sorption of mineral N on biochar isolates did not alter bulk soil mineral N availability, and this mechanism cannot account for reduced N₂O efflux in this study.

2 Woody biochar consistently reduced N₂O emissions relative to control treatments regardless of soil type or N fertilizer addition

One of the clearest patterns emerging from the aerobic incubation was the consistency of N₂O emissions in the biochar treatments relative to the control (Fig. 2). While only two fertilization levels and four soil types were tested, neither fertilization nor soil type significantly impacted this relationship, suggesting that soil properties or substrate availability may have a limited impact on the percent N₂O reductions from biochar amendments. This finding confirms the findings of Thomazini et al. (2015) who discovered a similar relationship between biochar and control N₂O across ten different soil types. Our study extends the trend to fertilized treatments with greater N₂O production. This implies the relative reduction achieved by the biochar may be driven more by biochar properties rather than by the soil properties.

Biochar’s ability to impact gaseous diffusion of N₂O from soil could provide one possible mechanism for the consistent percent N₂O reductions across treatments. Diffusion of N₂O from soils is an often-overlooked process which could reduce N₂O emissions independent of N₂O production (Blagodatsky & Smith, 2012). Harter et al. (2016) demonstrated biochar’s ability to entrap N₂O in water-filled pores under anaerobic soil conditions. Similarly, under a range of aerobic WFPS in sterilized soils, Quin et al. (2015) showed significant sorption of N₂O to biochar treated soils. This may serve as a temporary mechanism to limit N₂O efflux by slowing diffusion and increasing the likelihood of N₂O being fully reduced to N₂ by denitrifying microbial communities within anaerobic microsites. Cornelissen et al. (2013) developed N₂O sorption isotherms for similar woody biochars showing their large N₂O sorption capacity, well above the amount of N₂O produced in this incubation. While other soil physical processes likely limit biochar’s full N₂O sorption potential in soils, this may provide a possible explanation for the consistent trend across soil type and fertilization observed in these aerobic incubations.
3 Biochar may reduce N$_2$O by aerating soil under moderately anaerobic conditions

Biochar treatments may also have differing impacts on N$_2$O production as soils transition from predominantly aerobic to anaerobic conditions. Woody biochar is characterized by containing residual macropores which may be more effective at retaining air-filled pores than other soil pores (Sun et al., 2012), despite in this study biochar pores only contributed 3.8% to the total porosity of the soil. Such biochar residual macropores exhibit much lower pore connectivity than bulk soil pores thus may more readily retain air-filled pore space reducing the prevalence of anaerobic microsites within the soil matrix (Sun et al., 2013; Schnee et al., 2016). Biochar porosity can retain air even when soils are fully saturated due to hydrophobic effects depending on biochar functional groups (Gray et al., 2014). This may have potential to decrease denitrification rates and subsequent N$_2$O production if biochar helps aerate the soil.

The fertilized CO soils incubated across a soil moisture gradient (E2) demonstrated that soil moisture significantly impacts biochar’s N$_2$O mitigation potential and the greatest potential is at moderately anaerobic soil moisture levels (Fig. 8). In the control soils, N$_2$O production from the moderately anaerobic and anaerobic treatments was far greater than the dry and aerobic treatments, indicating greater N$_2$O contributions from anaerobic process such as denitrification (Bateman & Baggs, 2005; Linn & Doran, 1984). The 86.9% reduction in N$_2$O efflux in the moderately anaerobic biochar treatments could be explained by a biochar shifting conditions from moderately anaerobic to a more aerobic soil environment where denitrification rates would be reduced. The higher NO$_3^-$ concentrations in biochar treatments at day 30 (Fig. 6b) may also suggest decreased NO$_3^-$ consumption through denitrification. The NH$_4^+$ concentration in the anaerobic treatments provides further evidence that biochar may help aerate the soils. The control soil retained 23% of the NH$_4^+$ added (Fig. 6a), possibly due to the limited O$_2$ availability for nitrification, while in the biochar treatments all the NH$_4^+$ was consumed. The impact of biochar on soil moisture distribution and redox dynamics within the soil show large potential to influence N$_2$O production. Future research must consider these dynamics especially when modeling field condition with variable soil redox.

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Supporting Information

Additional Supporting Information may be found online in the supporting information tab for this article:

Table S1. Mean value (avg) and standard error (SE) of soil properties for each treatment sampled at the end of the 60-day incubation, including soil organic carbon (SOC), total nitrogen (TN), and soil pH.

Figure S1. Biochar isolate mineral nitrogen (N) from the biochar soils separated by mineral N type (ammonium (NH4+), Fig. S1a; nitrate (NO3−), Fig. S1b) across the four soil moisture regimes (dry (D), aerobic (A), moderately anaerobic (MAn), anaerobic (AN); see text for details) measured at the end of the 28-day, fertilized, CO soil incubation (E2). Error bars display standard error.