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Biochar effects on soil organic carbon sequestration and acidity amelioration persist after 10 years

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Intensive cropping and long-term ammoniacal nitrogen (N) fertilization have degraded soil health in eastern Oregon dryland wheat systems, leading to soil acidification and declining soil organic carbon (SOC) stocks, which poses a critical threat to sustainability. This study assessed the impacts of a one-time biochar application on soil acidity, SOC sequestration, and nutrient dynamics over 10 years in a winter wheat–spring pea rotation. Biochar, derived from forest waste and applied only once in 2013 at rates of 11.2, 22.4, and 44.8 t ha^{-1}, was evaluated against both non-amended control plots and plots receiving nitrogen fertilizer alone. Key soil properties, including pH, SOC, labile carbon (POXC), cation exchange capacity (CEC), electrical conductivity (EC), nutrient concentrations, and mineralization rates, were measured. Results showed biochar significantly increased soil pH by up to 0.9 units, with improvements persisting for a decade, particularly at higher rates. Elevated pH positively correlated with improved CEC, indicating enhanced nutrient retention and better macro/micronutrient availability (Zn, Ca, Mg, K), reducing Fe solubility. Biochar instantly increased SOC stocks by 95–207% and maintained the stocks for more than 10 years, demonstrating long-term persistence, particularly at higher application rates. Biochar effectively maintained a higher labile carbon content (POXC), although a declining POXC/SOC ratio suggested a shift to more stabilized carbon pools. Mineralization changes were moderate, with non-significant increases in CO_2 efflux at higher biochar rates and no consistent net N mineralization trends, suggesting limited direct stimulation of microbial N cycling. Overall, a single alkaline biochar application provided sustained, long-term benefits, playing a dual role in mitigating acidity and enhancing carbon sequestration, thereby supporting a sustainable strategy for restoring soil fertility and ecosystem function and strengthening dryland agroecosystem resilience.

KEYWORDS
acidification, dryland wheat system, greenhouse gas, labile carbon, silt loam, soil pH

1 Introduction

Intensive cropping and chemical inputs are increasingly challenging the sustainability of agricultural soils worldwide (Paramesh et al., 2023). In eastern Oregon, the long-established dryland winter wheat (*Triticum aestivum*)–summer fallow (WW–SF) cropping system has significantly degraded soil health. Limited precipitation, only 229–305 mm in most of the region, has led to the widespread adoption of WW–SF (Schillinger and Papendick, 2008). In this system, land is cropped for 1 year and left fallow the next to conserve winter precipitation, which, when combined with growing-season precipitation, typically produces reliable wheat yields (Ghimire et al., 2017). However, growing a single crop over 2 years produces insufficient biomass for SOC accretion under both conventional tillage (CT) and no-till (NT) WW-SF systems (Awale et al., 2022). Decades of low biomass input, exacerbated by CT practices, have contributed to the loss of up to 60% of soil organic carbon (SOC) in the topsoil layers, thereby compromising soil structure, water retention, and overall soil health and fertility (Rasmussen and Parton, 1994; West and Post, 2002; Machado, 2011; Bista et al., 2019). In addition, eastern Oregon's semiarid climate, with <330 mm of precipitation, limits SOC accumulation. Crop residue production is typically low, and most precipitation occurs in winter when wheat plants are dormant.

Concurrently, soils in the dryland wheat-producing regions of eastern Oregon have gradually become more acidic over time due to the continued application of ammonium-based nitrogen (N) fertilizers, including urea, ammonium nitrate, and ammonium sulfate. These fertilizers contribute to soil acidification by releasing hydrogen ions (H^+) during the nitrification process, where ammonium ($NH_4{}^+$) is oxidized to nitrate ($NO_3{}^-$; Meng et al., 2019; Dincá et al., 2022). Historically, most native prairie soils in the US Inland Pacific Northwest (iPNW) have had neutral to near-neutral pH levels, typically ranging from 6.5 to 7.2. However, decades of N fertilizer application have significantly acidified these soils, reducing the pH of the top foot of soil to below 5.2, which is nearer to the critical threshold for wheat production (Mahler, 1986; McFarland and Huggins, 2015). In addition to fertilizer use, soil acidification is exacerbated by leaching and plant uptake of basic cations (Ca, Mg, K, Na), acid rain, and decomposition of soil organic matter (SOM; Tian and Niu, 2015; Ibrahim et al., 2024). Increased soil acidity reduces crop productivity by limiting the availability of essential plant nutrients, increasing the solubility of toxic plant elements such as Al and Mn, and increasing the incidence of winter kill and disease (Awale et al., 2018). As soil productivity decreases, more artificial fertilizer is required to maintain high crop yields. High fertilizer rates lead to groundwater and air pollution (Snyder et al., 2009; Strebel et al., 1989). Despite the emerging threat of soil acidification, region-specific research and management guidelines remain limited. Growers lack science-based strategies to address declining pH, highlighting the urgent need for practical, lasting, and cost-efficient solutions that safeguard crop productivity.

Biochar, a carbon-rich product derived from biomass pyrolysis, has emerged as a promising soil amendment to counter these degradation processes (Laird et al., 2009). Its inherent recalcitrant carbon (C) structure contributes to sustained SOC sequestration and enhances soil physical properties and nutrient retention (Chagas et al., 2022; Kabir et al., 2023). Its intrinsic stability enhances SOC levels and acts as a robust climate change mitigation tool by sequestering C that would otherwise contribute to greenhouse gas emissions (Woolf et al., 2010). Moreover, depending on the feedstock and pyrolysis conditions, biochar can exhibit alkaline properties capable of neutralizing acidic soils (Singh et al., 2010). Unlike conventional lime applications, which often require reapplication every 2–3 years to sustain their pH-balancing effect, the recalcitrant nature of biochar has the potential to deliver long-lasting amelioration of soil acidity. This dual capacity for long-term C sequestration and lasting pH stabilization positions biochar as a promising amendment for enhancing both soil health and environmental resilience. Studies have demonstrated that when applied at appropriate rates, biochar can increase SOC and elevate soil pH, offsetting the deleterious effects of the WW-SF cropping systems as well as prolonged ammoniacal fertilizer use (Carr and Ritchie, 1993; Machado et al., 2018; Bista et al., 2019). This dual ability to provide long-term SOC sequestration along with lasting pH buffering positions biochar as a promising amendment for promoting both soil health and environmental resilience in regions like eastern Oregon.

This study aims to demonstrate that alkaline biochar's ability to enhance SOC stocks and ameliorate soil acidity can persist for at least a decade. In this paper, we assess the long-term potential of biochar to address two critical challenges in the Walla Walla silt loams of eastern Oregon: (1) the replenishment of SOC severely depleted by the WW–SF cropping system and (2) the mitigation of soil acidity caused by the prolonged use of ammoniacal N fertilizers. By investigating the mechanisms underlying biochar's effectiveness, ranging from its role in enhancing the biochemical stabilization of organic matter to its capacity for mitigating acidic conditions, we aim to provide insights into optimal application strategies. Our discussion integrates findings from recent field and laboratory studies to establish the viability of biochar as a sustainable amendment and its potential to restore soil functionality in this unique agroecosystem.

2 Materials and methods

2.1 Site description and experimental design

The study was conducted on an ongoing soil biochar amendment long-term experiment (BC-LTE) established in 2013 on a nearly-level (0–1% slope) Walla Walla silt loam soil (coarse-silty, mixed, mesic Typic Haploxeroll; Soil Survey Staff, 2014) at the Columbia Basin Agricultural Research Center (CBARC) near Pendleton, Oregon, USA (45 °42′ N, 118 ° 35′W). The site is characterized by a wide variation in mean monthly air temperatures, from −1 °C in January to 21 °C in July, with 135–170 frost-free days (June–September). The long-term annual precipitation at the site from 1930 to 2010 was 418 mm, with 70% of this amount received from September to April.

The experimental design was a split-plot with crops as the main plot factors and biochar and fertilizer combinations as subplot factors, replicated three times (Figure 1). Treatments were randomized in each replication. The plot size was 3 × 6 m. Wheat and pea were rotated each year, and each phase was present each year to facilitate annual data collection. The biochar used in this study contained 90% C, 0.18% N (C: N ratio of 500), and a strongly alkaline pH of 10.6. The biochar was applied only once to field plots in a winter wheat-spring pea (*Pisum sativum* L.) rotation in 2013 at rates of 0 (T1), 0 (T2), 11.2 (T3), 22.4 (T5),

FIGURE 1
Experimental design for evaluating biochar and nitrogen (N) fertilizer effects on pea–wheat rotation system in eastern Oregon. **(A)** Schematic layout of the split-plot field experiment showing the randomized distribution of five treatments (T1–T5) across two crop phases (pea and wheat) and three blocks (I–III). **(B)** Aerial image of the pea-phase plots outlined in red, with treatment positions labeled for reference. Image obtained from Google Earth.

TABLE 1 Biochar and fertilizer treatments.

Treatment (T)	Wheat		Spring pea	
	Biochar (t ha^{-1})	Fertilizer (N kg ha^{-1})	Biochar (t ha^{-1})	Fertilizer (N kg ha^{-1})
1	0	18	0	18
2	0	94	0	18
3	11.2	94	11.2	18
4	22.4	94	22.4	18
5	44.8	94	44.8	18

Biochar was applied once at the start of the experiment in 2013 to both wheat and pea phases, while nitrogen (N) fertilizer was applied annually throughout the study period.

and 44.8 (T5) t ha^{-1} (Table 1). The soil at the experimental site was initially acidic, with a mean pH (1:1) of 5.1 and a moderate organic matter content of 1.8%. T1 and T2 served as controls, with no biochar added, but received 18 and 90 kg N ha^{-1}, respectively, during the wheat phase to evaluate the effects of low and high N, in the absence of biochar (Table 1). Treatments T3, T4, and T5 also received 90 kg N ha^{-1} during the wheat phase. Under the pea phase, all treatments received 18 kg N ha^{-1} as the pea was expected to fix atmospheric N for its growth, leaving some N for the subsequent wheat crop. Fertilizer N treatments are applied every year.

2.2 Soil sampling and analysis

The soil was sampled at the 0–10, 10–20, and 20–30 cm depth intervals using 3.6 cm-diameter cores collected with a tractor-mounted hydraulic probe (Giddings Machine Company, Inc., Windsor, CO) after harvest in 2016 and 2024.

Total carbon (TC) and nitrogen (N) concentrations in the 2016 samples were determined using a LECO CN628 analyzer (LECO Corp., St. Joseph, MI, USA) via dry combustion at 950 °C. Previous studies have confirmed the absence of inorganic carbon in the 0–15 cm soil layer at the experimental site (Ghimire et al., 2015), and pH values below 6.7 across all samples further support the assumption that total carbon (TC) reflects soil organic carbon (SOC). Soil pH was measured in a 1:2 soil-to-solution ratio using 0.01 M $CaCl_2$. Five grams of air-dried soil (<2 mm) were mixed with 10 mL of extractant, and the pH was measured using an Orion Star A215 pH/conductivity benchtop meter (Thermo Fisher Scientific Inc., Beverly, MA, USA), following the method described by Ghimire et al. (2015). Bulk density was determined from soil cores sectioned into 10-cm increments, oven-dried at 105 °C, and weighed following the method described by Blake and Hartge (1986).

In 2024, TC and N were analyzed at the Oregon State University Soil Testing Laboratory using an Elementar Vario Max CNS analyzer via dry combustion. Soil pH was measured in a 1:1 soil-to-water suspension using a Hanna HI5522 benchtop meter. Permanganate oxidizable carbon (POXC) was determined following a modified protocol based on Weil et al. (2003) and Culman et al. (2012). Briefly, soil samples were mixed with

0.02 M KMnO4, shaken, and absorbance was measured at 550 nm using a UV-VIS spectrophotometer (Thermo Fisher Scientific Inc., Madison, WI, USA). Water-stable aggregate (WSA) stability was assessed using 20 g of air-dried soil aggregates (<2 mm). Aggregates were evenly spread on a 0.25 mm sieve and subjected to a simulated 5-min rainfall using a Cornell Sprinkle Infiltrometer (Cornell Soil Health Lab, Ithaca, NY, USA). For nutrient analysis, the soil was extracted with a Mehlich 3 solution containing acetic acid, ammonium nitrate, ammonium fluoride, nitric acid, and EDTA, using a standardized soil-to-extractant ratio. The suspension was thoroughly shaken, filtered, and analyzed for phosphorus (P), potassium (K), calcium (Ca), and magnesium (Mg) using an Agilent 5110 ICP-OES. The instrument was calibrated with certified standards, and quality control included blanks and replicates. Cation exchange capacity (CEC) was estimated by summing the exchangeable base cations from the Mehlich 3 extract and converting these to standard units for soil analysis. Soil microbial respiration (CO_2) was determined following a 24-h incubation of moist soil using an infrared gas analyzer (Haney et al., 2008). Briefly, a 40 g dry soil sample was moistened to 50% of its pore volume and incubated in an airtight jar for 24 h. Following incubation, gas samples were collected and analyzed using an infrared gas analyzer. Nitrogen mineralization was measured using 7-day anaerobic incubation (Drinkwater et al., 2015). Briefly, 7.5 g of soil sample is extracted with 2 M KCl to estimate NO_3 and NH_4, and then incubated anaerobically at 50% pore volume moisture for 7 days, covered with polythene film. After 7 days, the sample was thoroughly mixed and extracted with 2 M KCl for NO_3 and NH_4. Nitrogen mineralization was determined as the difference between NO_3 and NH_4 in incubated and unincubated soils, divided by the number of incubation days.

2.3 Yield estimate

Wheat yields were determined at physiological maturity from four 1 m^2 bundle samples per plot. A one-meter square frame was thrown at random onto the plots, and the wheat and peas inside the square frame was cut at the crown, and the grain from the sample was threshed and weighed. The grain yield per hectare was then determined by simple proportions as follows:

$$\frac{x\,(kg)}{4\,(m^2)} * \frac{1\,(Mg)}{1000\,(kg)} * \frac{10,000\,(m^2)}{1\,(ha)}$$

where x is the total wheat or pea grain (kg) from the four harvested bundles.

Wheat yields reported in the experiment were reliable from 2014 to 2017. From 2018 onwards, the plots were heavily infested by gophers, compromising crop stands and preventing reliable yield estimation. Therefore, yield data from 2018 onward are not reported in this study. Efforts to mitigate gopher damage are currently underway. Although we will discuss crop yield responses to biochar amendment, the main focus of this paper is on the effect of biochar on ameliorating soil acidity and SOC sequestration.

2.4 Statistical analysis

To evaluate treatment effects, both one-way and two-way analyses of variance (ANOVA) were performed. Data normality was assessed using the Shapiro–Wilk test, and when significant differences were identified, Tukey's HSD test was used for *post hoc* comparisons at a significance level of $\alpha = 0.05$. All analyses were conducted using StatGraphics Centurion. A two-way ANOVA was applied to examine C and N mineralization patterns and to assess whether wheat and pea crops responded differently. This analysis included testing the interaction between treatment and crop type (Treatment × Crop) on soil respiration. Statistical analyses and graphs were generated using R software (version 3.1.3) and OriginPro software (Origin 2024b, Origin Lab Corporation). Principal component analysis (PCA) was performed and visualized using CANOCO for Windows (version 4.5; Microcomputer Power, Ithaca, NY, USA) to explore relationships between soil properties and treatments. The significance of variable contributions to the PCA ordination was assessed using a Monte Carlo permutation test.

3 Results

3.1 Soil pH dynamics following biochar application

Soil pH responded significantly to biochar additions. Despite the use of different pH determination methods in 2016 and 2024, the trends in pH responses were similar. In 2016, 3 years after treatment initiation, there were clear differences in pH among treatments (Figure 2). Treatments T1 and T2 maintained significantly lower pH values than T5 ($p < 0.0001$), while no significant differences were observed between biochar-amended treatments at T3 (11.2 t ha $^{-1}$) and T4 (22.4 t ha $^{-1}$) application rates ($p = 0.32$). The highest mean pH values occurred in treatments T4 and T5 (pH 6.0 ± 0.72 and 5.94 ± 0.87, respectively), contrasting with the lowest values in T1 and T2 (pH < 5.2). The soil pH trend followed the order: T5 > T4 > T3 > T2 > T1, reflecting the positive impact of biochar amendments on acidity amelioration. A similar trend was observed in 2024, where the pH increases remain clearly detectable even 10 years after application. This effect was especially pronounced in treatments T4 and T5, indicating long-term soil alkalinization effects of biochar. On average, biochar applications increased soil pH by 0.38, 0.54, and 0.81 units at rates of 11.2, 22.4, and 44.8 t ha^{-1}, respectively, relative to T2 (no biochar, high N).

These pH changes were predominantly observed in the 0–10 cm soil depth, highlighting the localized effect of biochar in the surface soil layer where it was applied.

3.2 Soil organic carbon and nutrient responses to biochar

Biochar amendments significantly increased SOC concentrations and stocks, particularly at higher application rates in 2016 and 2024 (Figure 3). SOC stocks increased by 95%,

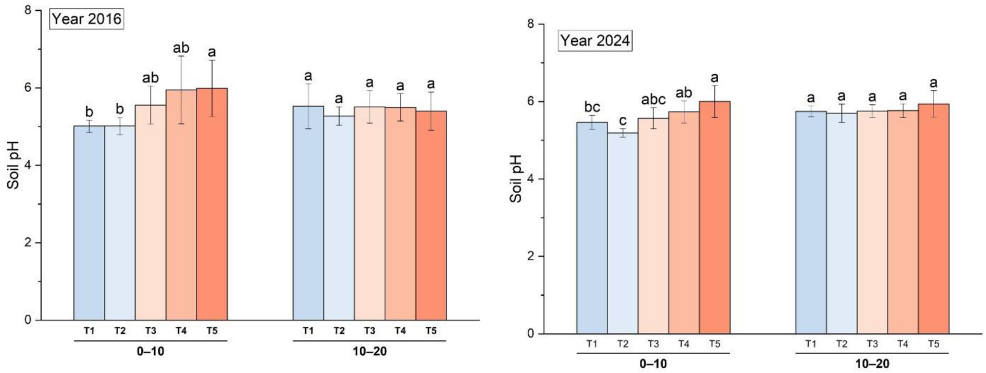

FIGURE 2
Short- (2016) and long-term (2024) effects of biochar treatments on soil pH at two depths (0–10 cm and 10–20 cm), 3 and 11 years after experiment establishment in 2013.Treatments: T1, control [no biochar, low nitrogen (N)]; T2, (no biochar, high N); T3–T5, biochar applied at 11.2, 22.4, and 44.8 t ha^{-1}. N was applied in T2–T5 at 90 kg ha^{-1} for wheat and 18 kg ha^{-1} for spring pea. Values are means ± SE ($n = 6$). Different letters within each depth indicate significant differences among treatments (Tukey's HSD, $p \leq 0.05$).

FIGURE 3
Soil organic carbon (SOC) stocks (t C ha^{-1}) at two depths (0–10 cm and 10–20 cm) for five treatments (T1–T5) in 2016 and 2024, following 3 and 11 years of management since experiment establishment in 2013. Treatments: T1, control [no biochar, low nitrogen (N)]; T2, (no biochar, high N); T3–T5, biochar applied at 11.2, 22.4, and 44.8 t ha^{-1}. N was applied in T2–T5 at 90 kg ha^{-1} for wheat and 18 kg ha^{-1} for spring pea. Values are means ± SD ($n = 6$). Different lowercase letters (green bars) indicate significant differences among treatments at 0–10 cm, and uppercase letters (red bars) indicate differences at 10–20 cm (Tukey's HSD, $p < 0.05$).

144%, and 207% in T3, T4, and T5, respectively, compared to T1. Although C stock seemed lower after 10 years, the overall trend in C gains persisted in T3, T4, and T5. The lower rates in 2024 than in 2016 could be due to the accuracy of C estimation using the Elementar. Notably, T2, which received a higher N application rate than T1 (both with no biochar), showed lower organic C stock than T1.

High biochar application rates significantly increased soil nutrient retention capacity, as evidenced by marked increases in cation exchange capacity (CEC). CEC measured across treatment groups (Table 2) showed a significant increase at high biochar rates, with values reaching 17.7 meq 100^{-1} g at a biochar application rate of 44.8 t ha^{-1} in treatment T5 ($p < 0.0003$). This indicates a substantial improvement in CEC over the non-biochar amended soils in T1 and T2 and also exceeds the CEC observed in T3, where a lower biochar rate of 11.2 t ha^{-1} was applied.

Total N concentrations showed no significant differences between treatments and the control (T1), which received no fertilizer N inputs. Accordingly, the higher C/N ratios observed in T4 and T5 at 0–10 cm-—18.8 and 23.8, respectively—reflect increases in SOC only (Table 2). Furthermore, biochar generally elevated POXC levels, notably in T4 and T5 (Table 2). POXC content increased significantly with higher biochar rates. However, no significant differences were detected among treatments in the POXC/SOC ratio. Notably, while biochar application tends to increase SOC, it may simultaneously reduce POXC, particularly in T4 and T5, resulting in a relative decrease in the POXC/SOC ratio. Biochar

TABLE 2 Soil properties (mean ± SD) at two depths (0–10 cm and 10–20 cm) across five treatments (T1–T5), including soil organic carbon (SOC), total nitrogen (TN), permanganate oxidizable carbon (POXC), POXC/SOC ratio, water-stable aggregates (WSA), and cation exchange capacity (CEC).

Treatment (T)	Depth (cm)	SOC (g kg^{-1})	TN (g kg^{-1})	POXC (mg kg^{-1})	POXC/SOC	WSA (%)	CEC (meq/100 g)
T1	0–10	14.2 ± 4.6ab	1.2 ± 0.2a	346 ± 33bc	26.4 ± 8.5a	24.4 ± 4.9a	11.4 ± 1.9bc
T2		12.3 ± 2.1b	1.1 ± 0a	344 ± 24c	28.6 ± 4.1a	27.8 ± 5.6a	10.5 ± 1.4c
T3		17.7 ± 6.4ab	1.2 ± 0a	372 ± 18abc	23.7 ± 9.4a	25.6 ± 3.5a	13.2 ± 2bc
T4		22.5 ± 11.3ab	1.2 ± 0a	394 ± 26ab	22.5 ± 12.6a	24.8 ± 3.6a	15.2 ± 3ab
T5		28.5 ± 15.1a	1.2 ± 0a	408 ± 30a	19.6 ± 12.7a	27.9 ± 3.9a	17.7 ± 3.8a
T1	10–20	13.5 ± 4.9a	0.9 ± 0a	282 ± 23a	23.1 ± 7.8a	24.9 ± 6.2a	12.5 ± 1.4a
T2		14.2 ± 9.6a	1.0 ± 0a	286 ± 34a	25.5 ± 10.7a	20.5 ± 3.9a	12.6 ± 1.2a
T3		11.5 ± 2.9a	1.0 ± 0a	286 ± 12a	26.1 ± 6a	21.9 ± 5.7a	13.6 ± 1.9a
T4		15 ± 7.7a	1.1 ± 0a	269 ± 29a	21.7 ± 9.1a	25.1 ± 8.7a	12.8 ± 1.1a
T5		16.1 ± 7.7a	1.0 ± 0a	270 ± 18a	20.2 ± 8.9a	20.6 ± 6a	13.5 ± 1.3a

Samples were collected in 2024, 11 years after the experiment establishment (2013). Different letters indicate significant differences within each depth (Tukey test, $p < 0.05$).

application also had no significant effect on WSA compared to the control (T1) and T2 at either 0–10 cm or 10–20 cm soil depth.

3.3 Relationships between treatments and other soil properties

Biochar application resulted in persistent and dose-dependent improvements in soil chemical properties more than a decade after a single application. The strongest responses were observed for base cations (Ca, Mg, K), P, Zn, and CEC, all of which increased significantly with higher biochar rates. In contrast, EC, Cu, and Fe remained largely unchanged (Table 3). Principal Component Analysis (PCA) was conducted to evaluate the relationships between treatment factors and soil properties in the 0–10 cm (Figure 4A) and 10–20 cm (Figure 4B) soil layers. At the 0–10 cm depth, PCA showed that treatments explained 40.4% of the total variation in soil variables, with PC1 and PC2 accounting for 69.0% and 16.0% of this variation, respectively. The analysis clearly shows the role of biochar in modulating soil properties, primarily through its influence on pH and nutrient availability. CEC, pH, and nutrient concentrations were positively correlated with T5 (Figure 4A). Soil pH, in particular, exhibited strong correlations with multiple soil properties and was notably high under T4 and T5. A positive correlation between EC and pH was observed in biochar-amended soil. This relationship was also associated with a consistent increase in soluble cations as pH rises. Positive correlations were observed between EC and key macronutrients (N, P, and K), as well as base cations (Ca, Mg, Mn, and Cu), indicating enhanced nutrient availability under biochar-amended treatments. Higher biochar application rates in treatments T4 and T5 were associated with increased Zn concentrations throughout the 0–20 cm soil depth. Conversely, lower soil pH in T1 and T2 plots with no biochar, which remained more acidic, was associated with higher Fe concentrations in the 10 cm depth profile.

3.4 Influence of biochar on CO$_2$ efflux and N mineralization

Soil C and N mineralization rates were measured at two depth intervals (0–10 cm and 10–20 cm) across the five treatments (T1 to T5), as shown in Figures 5, 6. These measurements were conducted in 2024, 11 years after the experiment was established. No initial baseline data are available. The results indicate that CO$_2$ emissions in soils amended with higher biochar rates ranged from 16.85 to 23.98 μg CO$_2$-C/g dry soil/day, with the highest emissions observed in T5 although the emissions were not significantly different among treatments (Figure 5). The N-added non-biochar treatment (T2) showed the lowest emission rate at 16.85 μg CO$_2$-C/g dry soil/day. The biochar contributed only a minor increase in CO$_2$-C emissions, and the magnitude of this increase in CO$_2$ efflux was minimal compared with that in the non-amended soil. The most pronounced increases in CO$_2$-C emissions were observed in the biochar-treated plots. Compared to T2, treatment T5 exhibited a 42% higher cumulative CO$_2$-C emission, while treatments T3 and T4 showed increases of approximately 30%. However, these increases were not statistically significant ($P > 0.05$) compared to their respective control treatments (Figure 5). The higher respiration rates were confined to the 0–10 cm soil depth. In contrast, the 10–20 cm layer showed no significant differences, with values ranging from 5.66 to 8.56 μg CO$_2$-C/g dry soil/day.

The results indicate that biochar amendments did not produce a consistent or statistically significant effect on N mineralization rates across the different treatments (Figure 6). At the surface soil layer (0–10 cm), N mineralization rates varied between 0.259 and 0.420 mg N/kg soil/day, with T4 showing the highest rate and T2 the lowest. However, these differences were not significant, suggesting that the influence of biochar on N mineralization at this depth was limited or variable. At the 10–20 cm depth, the mean mineralization rates were considerably lower, ranging from 0.071 to 0.099 mg N/kg soil/day, with no significant differences among treatments within this depth interval. T1 showed the highest mean rate (0.099 ± 0.010 mg N/kg soil/day), while T2 had the lowest (0.071 mg N/kg soil/day). We did not separate wheat and pea crops to evaluate

TABLE 3 Soil chemical properties (0–10 cm depth) as affected by increasing biochar application rates under dryland conditions.

Treatment (T), biochar (t ha^{-1})	EC (dS/m)	Ca (mg/kg)	Cu (mg/kg)	Fe (mg/kg)	K (mg/kg)	Mg (mg/kg)	Mn (mg/kg)	P (mg/kg)	Zn (mg/kg)
T1 (0)	0.1 ± 0.1a	1,272 ± 231d	1.9 ± 0.4a	241 ± 15a	1,122 ± 101c	265 ± 63c	75.9 ± 8.8b	100 ± 7c	0.8 ± 0.3c
T2 (0)	0.1 ± 0.0a	1,121 ± 123d	2.0 ± 0.5a	259 ± 33a	1,130 ± 182c	238 ± 43c	77.5 ± 13.4b	111 ± 19c	0.9 ± 0.3c
T3 (11.2)	0.2 ± 0.0a	1,505 ± 254c	2.0 ± 0.2a	237 ± 38a	1,259 ± 123b	289 ± 58bc	87.3 ± 11.9b	131 ± 14b	1.8 ± 0.7b
T4 (22.4)	0.2 ± 0.1a	1,804 ± 443b	2.4 ± 0.6a	230 ± 35a	1,356 ± 163b	326 ± 59b	98.5 ± 23.9ab	150 ± 42b	3.5 ± 2.3ab
T5 (44.8)	0.3 ± 0.1a	2,150 ± 558a	2.4 ± 0.3a	217 ± 21a	1,460 ± 255a	387 ± 62a	115 ± 31a	187 ± 67a	5.1 ± 2.9a

Values represent treatment means ± standard deviation (n = 6). Different letters within a row indicate significant differences among treatments according to Tukey's HSD test ($p < 0.05$).

trends in mineralization because no significant interaction effects between crop types (wheat and pea) and treatments were found for either CO_2-C emission rates or N mineralization during the 24-h incubation period (Table 4). Specifically, the Treatment × Crop interactions on soil carbon respiration rate and net nitrogen mineralization were not significant ($P = 0.061$ and $P = 0.458$, respectively). This indicated that the mineralization responses were consistent across these crop types, permitting only main effects analysis. Biochar rates may offer the most agronomic benefit under the conditions of this study.

3.5 Biochar effects on grain yield

3.5.1 Wheat

Mean wheat yields across the 4 years increased from 4.277 t ha^{-1} in the unfertilized control (T1) to 4.436 t ha^{-1} with N alone (T2), representing an absolute gain of 0.159 t ha^{-1} (≈3.7%; Figure 7). Adding biochar with 90 kg N produced further gains: T3 = 4.615 t ha^{-1} (+0.179 t ha^{-1} vs. T2, ≈4.0%), and T4 = 4.834 t ha^{-1} (+0.219 t ha^{-1} vs. T3, ≈4.7%; Figure 7b). The highest biochar rate (T5 = 4.732 t ha^{-1}) did not increase mean yield relative to T4 and was slightly lower (−0.102 t ha^{-1}, ≈-2.1%). Standard errors of the 4-year means vary by treatment (T1 SE ≈ 0.294; T2 SE ≈ 0.027; T3 SE ≈ 0.122; T4 SE ≈ 0.137; T5 SE ≈ 0.359), indicating that some treatments, particularly T5, exhibit greater interannual variability (Figure 7a). Yearly values show this variability clearly (for example, wheat under T5 dropped to 3.579 t ha^{-1} in 2016 and rose to 5.105 t ha^{-1} in 2017).

3.5.2 Peas

Pea mean yields show a much stronger response to biochar when combined with the modest N rate (18 kg N ha^{-1}; Figure 7d). The unfertilized control (T1) averaged 1.283 t ha^{-1}, and N alone (T2) gave only a small increase to 1.309 t ha^{-1} (+0.026 t ha^{-1}, ≈2.0%). In contrast, adding biochar produced large gains: T3 (11.2 t ha^{-1}) averaged 1.937 t ha^{-1}, an absolute increase of 0.628 t ha^{-1} over T2 (≈48%). T4 (22.4 t ha^{-1}) averaged 1.887 t ha^{-1}, slightly below T3 (−0.050 t ha^{-1}, ≈-2.6%), and T5 (44.4 t ha^{-1}) averaged 1.782 t ha^{-1} (−0.105 t ha^{-1} vs. T4, ≈-5.6%). Standard errors for pea means are moderate and relatively consistent (T1 SE ≈ 0.118; T2 SE ≈ 0.053; T3 SE ≈ 0.120; T4 SE ≈ 0.115; T5 SE ≈ 0.037), indicating the large mean increase at T3 is not obviously driven by a single anomalous year (Figure 7c). There were no yields obtained in 2015 because of poor crop emergence.

3.5.3 Comparative interpretation

Two clear patterns emerge. First, biochar combined with mineral N produced larger incremental benefits than mineral N alone for both crops, but the magnitude differed: wheat showed modest incremental gains with each moderate biochar increment up to 22.4 t ha^{-1}, whereas peas showed

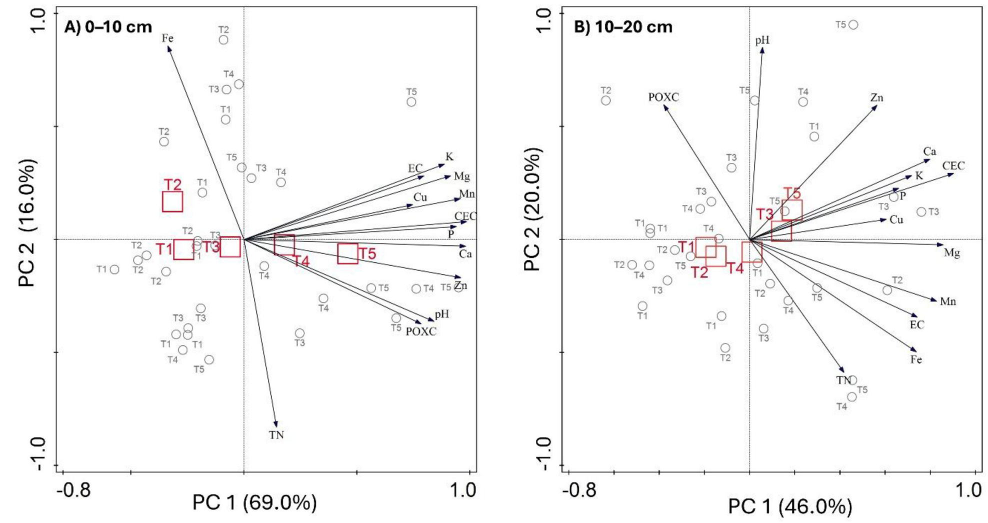

FIGURE 4
Principal component analysis (PCA) illustrating the relationships between soil nutrients and treatments at two depths (0–10 cm and 10–20 cm). Soil variables include electrical conductivity (EC), potassium (K), manganese (Mn), magnesium (Mg), phosphorus (P), calcium (Ca), cation exchange capacity (CEC), zinc (Zn), pH, permanganate oxidizable carbon (POXC), and total nitrogen (TN). Treatments: T1, control [no biochar, low nitrogen (N)]; T2, (no biochar, high N); T3–T5, biochar applied at 11.2, 22.4, and 44.8 t ha^{-1}. N was applied in T2–T5 at 90 kg ha^{-1} for wheat and 18 kg ha^{-1} for spring pea.

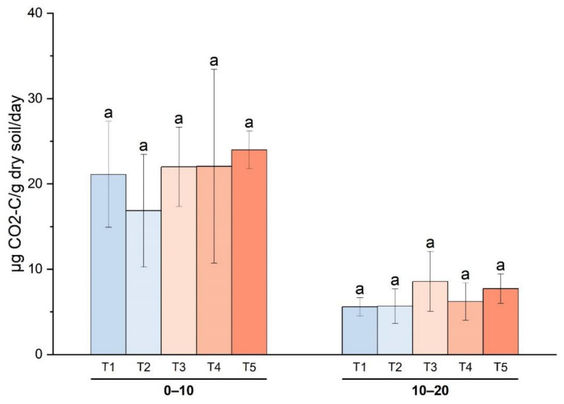

FIGURE 5
Effect of biochar treatments on soil CO_2 respiration (µg CO_2-C g^{-1} dry soil day^{-1}) after 11 years (2024) at two depths (0–10 cm and 10–20 cm) under five treatments: T1, control [no biochar, no nitrogen (N)]; T2, N only; T3–T5, biochar at 11.2, 22.4, and 44.8 t ha^{-1}, each with N added. N was applied at 90 kg ha^{-1} (wheat) and 18 kg ha^{-1} (spring pea). Values are means ± SE ($n = 6$). Different letters within each depth indicate significant differences among treatments (Tukey's HSD, $p \leq 0.05$).

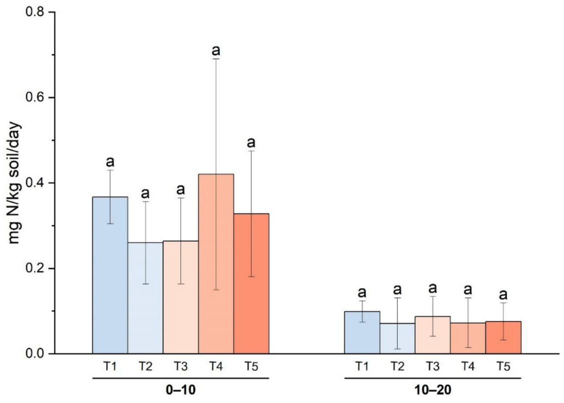

FIGURE 6
Effect of biochar treatments on nitrogen (N) mineralization rates (mg N kg^{-1} soil day^{-1}) after 11 years (2024) at two depths (0–10 cm and 10–20 cm) under five treatments: T1, control [no biochar, no nitrogen (N)]; T2, N only; T3–T5, biochar at 11.2, 22.4, and 44.8 t ha^{-1}, each with N added. N was applied at 90 kg ha^{-1} (wheat) and 18 kg ha^{-1} (spring pea). Values are means ± SE ($n = 6$). Different letters within each depth indicate significant differences among treatments (Tukey's HSD, $p \leq 0.05$).

a pronounced jump at the lowest biochar addition (11.2 t ha^{-1}) and no further benefit at higher rates (Figure 7). Second, the highest biochar rate (44.8 t ha^{-1}) did not improve mean yields for either crop and was associated with greater interannual variability for wheat. Taken together, these patterns suggest an optimal biochar window in this system (≈11–22 t ha^{-1}) rather than a monotonic positive response to increasing biochar.

4 Discussion

The results of this long-term study show that a single application of alkaline, forest-waste biochar produced sustained, practical improvements in soil health on Walla Walla silt loam soils in eastern Oregon's dryland wheat systems. In landscapes degraded by intensive cropping and prolonged ammoniacal N fertilization, where soil acidification and SOC losses are major constraints, biochar simultaneously mitigated acidity and

TABLE 4 Soil CO_2 respiration (μg CO_2-C g^{-1} dry soil day^{-1}) and net nitrogen (N) mineralization (mg N kg^{-1} soil day^{-1}) measured during a 24-h incubation for wheat and spring pea across five treatments (T1–T5).

Crop	Treatment	*n*	CO_2 respiration (μg CO_2-C/g dry soil/day)		N mineralization (mg N/kg soil/day)	
Wheat	T1	3	21.5 ± 9.5a		0.38 ± 0.05a	
Wheat	T2	3	12.6 ± 5.8a		0.22 ± 0.09a	
Wheat	T3	3	22.3 ± 1.7a		0.32 ± 0.11a	
Wheat	T4	3	29.3 ± 6.3a		0.54 ± 0.32a	
Wheat	T5	3	24.7 ± 2.3a		0.38 ± 0.04a	
Spring pea	T1	3	20.8 ± 2.5a		0.35 ± 0.08a	
Spring pea	T2	3	21.1 ± 4.6a		0.3 ± 0.1a	
Spring pea	T3	3	21.7 ± 7.1a		0.21 ± 0.07a	
Spring pea	T4	3	14.8 ± 11.1a		0.3 ± 0.19a	
Spring pea	T5	3	23.2 ± 2.3a		0.28 ± 0.21a	
			F	*P*	*F*	*P*
Treatment (T)			1.110	0.379	1.978	0.175
Crop (C)			0.606	0.445	1.224	0.332
T × C			2.689	0.061	0.945	0.458

Different letters indicate statistically significant differences (Tukey's HSD, $P > 0.05$). Interaction effects and *P* values for treatment × crop type (T × C) on soil respiration and N mineralization are also presented.

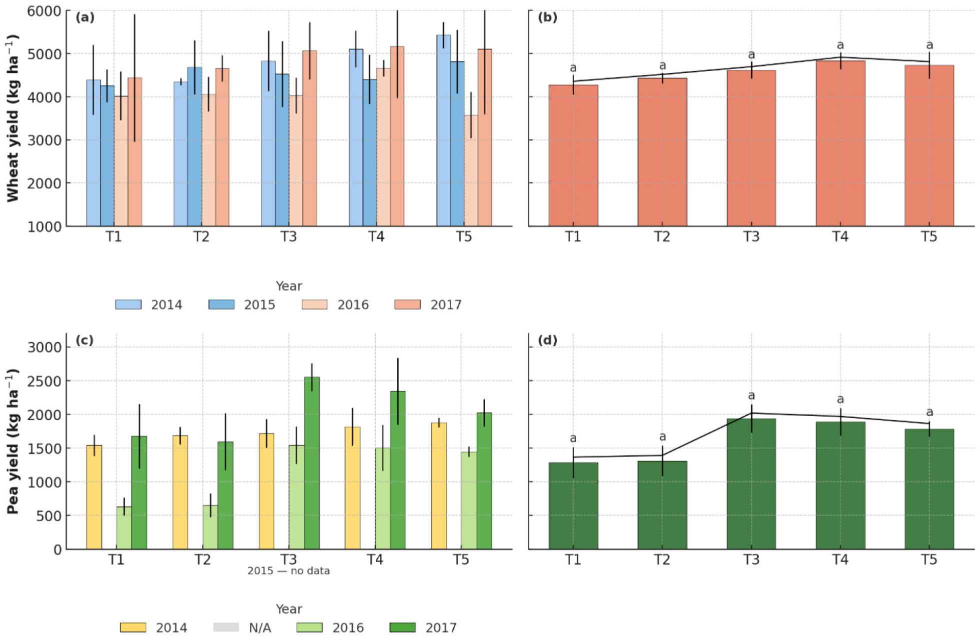

FIGURE 7
Wheat and spring pea yield (kg ha^{-1}) response to biochar across treatments T1–T5 during four consecutive years (2014–2017). **(a)** wheat treatment means± SE ($n = 3$) for each year (2014–2017), **(b)** wheat treatment means ± SE ($n = 12$) averaged across the 4-year period, **(c)** spring pea treatment means ± SE ($n = 3$) for each years (2014, 2016, and 2017), **(d)** spring pea treatment means ± SE ($n = 9$) averaged across those 3 years. Different letters indicate statistically significant differences among treatments (Tukey's HSD, $p < 0.05$).

increased SOC stocks. In addition, biochar enhanced nutrient retention and modestly affected soil respiration and nitrogen cycling, with secondary implications for crop performance. Together, these outcomes demonstrate biochar's dual capacity to restore soil fertility and bolster the resilience of semiarid dryland agroecosystems.

4.1 Soil pH dynamics and nutrient retention

A single application of alkaline forest-waste biochar in 2013 produced a pronounced and persistent increase in soil pH. Relative to the initial mean pH of 5.1, biochar raised pH by up to 0.9 units, with the largest and longest-lasting gains at the higher application rates (22.4 and 44.8 t ha^{-1}), and these increases persisted through the entire 10-year monitoring period. The pH response was concentrated in the surface 0–10 cm layer, reflecting the localized effect where biochar was incorporated. On a logarithmic scale, this change corresponds to a substantial reduction in acidity, consistent with biochar's liming capacity derived from its alkaline chemistry (Zhang et al., 2025). Our results confirm that initially acidic soils (mean pH 5.1) experienced notable amelioration across biochar-amended treatments, and that elevated pH levels, especially in T4 and T5, were sustained for a decade, highlighting biochar's long-term stability and its sustained influence on nutrient availability. While previous work has emphasized pH responses in sandy soils (Chintala et al., 2014; Idbella et al., 2024; Zhang et al., 2025), these findings demonstrate comparable benefits in silty Walla Walla silt loams. Rate-response patterns showed a plateau: pH did not differ significantly among the biochar amended treatments (11.2, 22.4, 44.8 t ha^{-1}), indicating diminishing returns once soil buffering capacity moderates further change (Yin et al., 2014; Wang et al., 2014; Tan et al., 2022). The highest pH values in T5 suggest that particular biochar–fertilizer combinations can maximize alkalinity (Ali et al., 2022). In contrast, persistently low pH in non-amended controls (T1, T2) underscores the limited capacity of unfertilized or conventionally fertilized treatments to reverse acidification. Sustained pH elevation from a single biochar application therefore has clear implications for nutrient retention: higher pH improved cation exchange capacity and reduced the solubility of toxic elements, thereby enhancing the availability of essential macro- and micronutrients in the root zone.

4.2 Enhancement of soil organic carbon stocks

Biochar amendments produced large and persistent increases in soil organic carbon (SOC), with the strongest responses at higher application rates. C stocks rose by 95%, 144%, and 207% in treatments T3, T4, and T5, respectively, demonstrating biochar's effectiveness as a long-term carbon sequestration agent. Three years after application, SOC in T3–T5 exceeded the N-only treatment (T2) by 73%—174%, underscoring the rapid and substantial C gains following a single biochar addition.

Although total C stocks declined somewhat by year ten, the higher-rate treatments (T4 and T5) maintained a clear positive trend, indicating sustained storage benefits from a one-time application. Instrument differences may partly explain some variation between the 2016 and 2024 measurements, since TC was measured on a LECO analyzer in 2016 and on an Elementar analyzer in 2024; nevertheless, the long-term elevation of SOC, including a 24% increase after 10 years following a single 11.2 t ha^{-1} application, points to durable biochar persistence in these soils. For context, other studies using repeated annual biochar applications have reported SOC increases up to 82.2% at rates up to 9.0 t ha^{-1} (Shi et al., 2021), highlighting that even a single, larger application can produce comparable long-term gains.

In contrast, the N-fertilized, non-biochar treatment (T2) showed a decline in soil C relative to the low-N control (T1), consistent with a priming effect in which elevated N availability stimulates microbial activity and accelerates the mineralization of stable SOC (Lu and Zhang, 2015). High N inputs can disrupt C/N balance and microbial stoichiometry, promoting decomposition of more recalcitrant soil organic matter and reducing total SOC (Bei et al., 2022). These contrasting outcomes emphasize that biochar can both add stable C directly and help protect existing SOC from N-driven losses.

4.2.1 Labile carbon (POXC) responses

Biochar application markedly increased permanganate-oxidizable carbon (POXC), especially in treatments T4 and T5, indicating an expansion of the labile or active carbon pool that fuels soil biological activity and nutrient cycling (Hurisso et al., 2016). The POXC rise likely reflects greater concentrations of readily oxidizable organic compounds, carbohydrates, simple sugars, and amino acids, originating from biochar itself and from biochar-stimulated microbial transformations (Hurisso et al., 2016).

Despite the absolute increase in POXC, the POXC/SOC ratio did not differ significantly among treatments, implying that labile C and total SOC increased in parallel after biochar addition. Over time, the relative share of labile C either stabilized or declined, consistent with progressive conversion of readily oxidizable material into more stabilized organic fractions (Culman et al., 2012; Wade et al., 2021). This pattern supports the view that biochar both supplies substrate for microbes and promotes formation of resistant carbon compounds, thereby shifting soil C toward a slower-cycling, more persistent pool, an effect that is often stronger for biochars produced at higher pyrolysis temperatures (Ouyang et al., 2016; Tian et al., 2016; Woolf et al., 2021).

By contrast, high rates of N fertilizer without biochar can accelerate organic C losses, via stimulated microbial decomposition or altered C turnover that raises POXC relative to SOC, thereby undermining long-term C sequestration (Zhou et al., 2025). These results suggest that combining biochar with balanced N management offers a promising strategy to increase stable soil C storage, support soil biological function, and mitigate the deleterious effects of N over-fertilization on SOC dynamics.

4.3 Soil properties responses to biochar

Principal component analysis (PCA) indicated that biochar substantially improved soil chemical properties by raising pH, increasing cation exchange capacity (CEC), and enhancing nutrient availability, changes that collectively support greater soil fertility and potential crop productivity (Aurangzeib et al., 2024). The positive correlation between electrical conductivity (EC) and pH under biochar treatments is consistent with increased soluble cation availability as soils become less acidic (Mia et al., 2014). This pattern coincided with higher concentrations of key macronutrients (N, P, K) and base cations in biochar-amended plots, suggesting improved nutrient retention and reduced fixation that favor plant uptake (Hossain et al., 2020; Nair et al., 2017). Notably, micronutrients such as Zn accumulated under the higher-rate treatments (T4, T5), indicating that biochar can enhance both carbon cycling and the availability of essential trace elements for crop growth (Xu et al., 2022). Improved nutrient and carbon status under biochar is also likely to support more robust microbial communities and more efficient nutrient cycling (Pokharel et al., 2020). By contrast, non-amended treatments (T1, T2) with lower pH showed elevated Fe concentrations, reflecting the greater Fe solubility typical of acidic soils and the associated risk of phytotoxicity or nutrient imbalance, further evidence that biochar mitigates acidity-driven micronutrient problems through pH regulation (Natasha et al., 2022).

Biochar did not significantly affect water-stable aggregate (WSA) stability in this study, a result that contrasts with reports of improved aggregation in some fine-silty soils (Wang et al., 2017; Thapa and Mowrer, 2024). These mixed outcomes underscore the importance of soil texture, baseline management, and methodological differences: long-term no-till systems often already exhibit inherent aggregate stability, which can mask additional structural benefits from biochar in silty soils. Measurement approaches and site-specific conditions therefore help explain why WSA responses to biochar vary across studies.

4.4 Biochar and soil mineralization processes

Growing evidence indicates that biochar can stimulate microbial respiration in amended soils, but these effects are typically moderate and often not statistically significant over common experimental timescales (Jaafar et al., 2015; Ventura et al., 2015; Liu et al., 2016). Our results align with this pattern: even at the highest biochar rates, we observed no significant increase in microbial respiration. Short-term pulses in microbial activity have been reported following biochar addition, attributed to an initial release of labile carbon, but such pulses are transient and usually subside as the labile fraction is consumed (Cheng et al., 2017). Over longer time scales, the recalcitrant, aromatic structure of biochar limits the supply of readily degradable C, thereby reducing its contribution to sustained microbial respiration (Lu et al., 2021).

The POXC/SOC response in our study supports this interpretation. High-rate biochar treatments showed elevated absolute POXC but a lower or stable POXC/SOC ratio, consistent with an initial boost in oxidizable C that progressively converts into more stable organic forms (Culman et al., 2012; Wade et al., 2021). Because early post-application measurements were not available, we cannot directly document any brief initial stimulation; nevertheless, the long-term POXC/SOC pattern is compatible with rapid consumption of labile inputs followed by stabilization.

Effects on net N mineralization were similarly variable and not statistically significant. Biochar can both immobilize N through adsorbing or binding inorganic forms and slowly release N over time, so its net impact depends on biochar properties and soil–microbe interactions (Prommer et al., 2014). In some contexts, biochar also stimulates microbial biomass and activity, thereby accelerating N turnover and partially offsetting immobilization (Ibrahim et al., 2020; Ahmad et al., 2021). The mixed outcomes observed here reflect that complexity.

Together, these findings emphasize that biochar's influence on C and N cycling is context-dependent, shaped by biochar chemistry, soil properties, and temporal dynamics, and that long-term monitoring is essential to resolve transient vs. persistent effects on microbial processes and nutrient fluxes.

4.5 Crop yield responses to biochar

The results of this 4-year field trial indicate that biochar applied at moderate rates (11.2–22.4 t ha^{-1}) together with mineral N improved crop performance relative to unfertilized controls and to N alone, with the strongest relative response observed in peas under low N (18 kg N ha^{-1}) and more modest incremental gains for wheat receiving 90 kg N ha^{-1}. These patterns are consistent with broader syntheses showing that biochar effects on crop productivity are context-dependent and often most positive when biochar is used in combination with fertilizer rather than as a sole amendment (Jeffery et al., 2011; Melo et al., 2022). Jeffery et al. (2011) reported a small but significant overall increase in mean yield from biochar across studies, while emphasizing substantial heterogeneity driven by soil, climate, crop, and management factors.

Meta-analyses have also shown that moderate application rates often deliver the best agronomic returns, whereas very high application rates yield inconsistent or null responses. Biederman and Harpole's synthesis found substantial variability in outcomes and noted that responses depend on baseline fertility and experimental conditions, supporting our observation that the 44.8 t ha^{-1} rate did not improve mean yields and was associated with greater year-to-year fluctuations (Biederman and Stanley Harpole, 2013). The pronounced pea response at the lowest biochar addition (11.2 t ha^{-1}) in our trial aligns with meta-analytic evidence that legumes and crops on lower-fertility soils can show larger relative gains from biochar, likely because biochar improves nutrient retention and soil physical properties that are limiting under low N inputs (Ye et al., 2020).

Mechanistically, the benefits observed at moderate biochar rates likely reflect improved soil physical structure, increased water-holding capacity, and enhanced nutrient retention, which together can increase fertilizer use efficiency and buffer plants against short-term stress. These mechanisms are well-documented in field syntheses and reviews and help explain why biochar paired with mineral N often outperforms either input alone (Jeffery et al., 2011; Ye et al., 2020). Conversely, the absence of additional benefit and the increased variability at the highest biochar rate may reflect diminishing returns or adverse effects such as temporary nutrient immobilization, pH shifts, or altered microbial community composition that reduce plant-available nutrients or disrupt beneficial interactions (Zhao et al., 2022; Lin et al., 2024). In our study, part of this variability at this rate is likely due to yield reductions in several plots that were affected by gopher disturbance during the growing season. Because this issue persisted in subsequent years and compromised the reliability of the yield measurements, more recent yield data could not be accurately estimated and are therefore not included in this analysis.

In summary, these findings reinforce the emerging consensus that biochar can enhance crop yields when matched to site conditions and fertilizer regimes, but that benefits are not guaranteed at very high application rates and are mediated by complex soil–plant processes (Jeffery et al., 2011; Biederman and Stanley Harpole, 2013; Ye et al., 2020).

5 Conclusion

This long-term field trial demonstrates that a single application of alkaline forest-waste biochar can deliver durable improvements in soil health and crop performance in a semiarid dryland wheat–pea rotation. Moderate biochar rates (≈11–22 t ha^{-1}) combined with appropriate mineral N produced the most consistent yield benefits, while the highest rate tested (≈44 t ha^{-1}) offered no clear agronomic advantage and increased interannual variability. Crucially, biochar's effects extended beyond short-term productivity gains: surface soil pH, cation exchange capacity, base cation availability, and soil organic carbon stocks remained elevated a decade after application, indicating persistent amelioration of acidity and long-term carbon sequestration potential.

These outcomes are directly relevant to sustainability and climate-smart food systems. By concurrently buffering soil acidity and increasing stable SOC, biochar can reduce the need for repeated lime and heavy fertilizer inputs, lower the risk of nutrient losses to air and water, and contribute to on-farm carbon storage. When combined with careful fertilizer management, biochar thus supports multiple sustainability objectives: maintaining or enhancing yields, improving nutrient use efficiency, and boosting soil carbon sinks that help reduce greenhouse gas emissions.

From an agro-ecosystem resilience perspective, the persistent improvements in soil chemical and physical properties suggest that biochar can strengthen system stability under the climatic and edaphic constraints typical of semiarid regions. Moderate biochar additions improved nutrient retention and soil buffering in the surface horizon, where crops access resources, thereby reducing vulnerability to episodic stress (drought, cold, or nutrient shocks) and supporting more reliable production across seasons. However, the lack of additional benefit at very high application rates and the observed variability underscore the importance of site-specific optimization rather than one-size-fits-all prescriptions.

For broader adoption and policy, these findings point to practical pathways: prioritize moderate, evidence-based biochar rates integrated with tailored fertilizer regimes; monitor soil chemical and biological indicators to guide adaptive management; and consider biochar as one component of a portfolio of climate-smart practices (cover cropping, residue retention, reduced tillage) that together enhance productivity and ecosystem services. Future work should quantify life-cycle greenhouse gas balances, economic costs and benefits at the farm scale, and longer-term agronomic outcomes across diverse soils and climates to refine recommendations and support scalable, resilient, and climate-smart agricultural systems.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

SM: Conceptualization, Investigation, Supervision, Visualization, Writing – original draft, Writing – review & editing, Funding acquisition, Methodology, Project administration, Resources, Validation. PR: Conceptualization, Investigation, Supervision, Visualization, Writing – original draft, Writing – review & editing, Data curation, Formal analysis, Software. SS: Investigation, Methodology, Validation, Writing – review & editing.

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Conflict of interest

The author(s) declared that this work was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Generative AI statement

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