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ORIGINAL PAPER

Residue and Biochar-based Amendments from Maize and Sorghum Cultivars Regulate CO_2 Emissions and Nutrient Mineralization

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Abstract

Soil fertility decline and climate change threaten sustainable crop production, creating a need for residue-based strategies that enhance nutrient cycling and soil carbon stabilization. This study investigated how feedstock and biochar produced from different maize and sorghum cultivars influence CO_2 emissions, nitrogen (N) and phosphorus (P) mineralization, and soil pH. An incubation experiment was conducted using feedstock (raw residues) and biochar derived from two maize (R201, SC701) and two sorghum (AS8, PAN8816) cultivars pyrolyzed at 350 °C and 650 °C. Amendments were applied to 100 g soil at 10 t C ha^{-1}, with a soil-only control. CO_2 emissions were trapped in NaOH and quantified over 120 days. A parallel destructive sampling experiment was set up to assess NH_4^+-N, NO_3^--N, extractable P, and soil pH at multiple time intervals. CO_2-C emissions increased rapidly at the beginning of the incubation, peaking between 21 and 42 days and declined thereafter. Feedstocks produced higher CO_2-C emissions than biochars for all cultivars, with PAN8816 feedstock showing the greatest increase (205% above the control). Biochar emissions decreased with increasing pyrolysis temperature. Nitrogen and phosphorus mineralization, as well as soil pH, also varied significantly among cultivars and pyrolysis temperatures. Relative to the unamended control, SC701-B650 increased soil NO_3^--N by 17%, while PAN8816-B350 increased both NH_4^+-N and NO_3^--N concentrations. Extractable P was highest in R201-B650 (maize) and AS8-B650 (sorghum). Soil pH generally increased in the order feedstock< B350 < B650. Cultivar-specific differences and pyrolysis temperature had stronger effects on CO_2-C emissions, nutrient mineralization, and soil pH than crop type. Biochars, particularly those produced at 650 °C, showed greater stability and lower CO_2-C emissions while enhancing nutrient availability. These findings demonstrate that selecting suitable cultivars and pyrolysis temperatures can improve soil fertility, support carbon sequestration, and contribute to sustainable residue management.

Keywords Biochar · Carbon sequestration · Crop residues · Nutrient mineralization · Soil fertility

1 Introduction

Soil fertility decline remains a major constraint to crop productivity, particularly in agricultural systems affected by monocropping, excessive fertilizer use, intensive tillage and other anthropogenic activities (Heerink 2005; Nord et al. 2022). These practices accelerate soil organic carbon (SOC) mineralization, increase CO_2 emissions, and contribute to SOC depletion and erosion. Sustainable soil management practices that enhance nutrient availability and maintain or increase SOC stocks are therefore essential. Organic inputs such as crop residues, manures, and organic wastes are widely recognized for improving soil fertility (Powlson et al. 2012). Globally, crop residues represent the most abundant organic resource, with cereal residues alone contributing approximately 74% of the 6 billion tons produced annually (Lal 2005). These residues contain essential nutrients including C, N, P, K and micronutrients, yet their agronomic value depends strongly on biochemical composition particularly C, N, P, lignin content, and C: N ratio (Kaleem Abbasi et al., 2015; Schmatz et al. 2017). Residues with relatively low C: N ratios (<25:1), typically characterised by higher nitrogen content and lower concentrations of lignin and polyphenols, are more likely to decompose readily and result in net nitrogen mineralization. In contrast, residues with wider C: N ratios (>25:1), lower nitrogen concentration,

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and higher proportions of recalcitrant biochemical compounds tend to decompose more slowly and may lead to temporary nitrogen immobilization (Aher et al. 2017; Partey et al. 2019). Although C: N ratio is a key determinant of residue decomposition, it does not operate in isolation, residue decomposition and nutrient release are also regulated by other biochemical quality parameters, including lignin, polyphenols, cellulose fractions, and associated stoichiometric ratios such as lignin: N, polyphenol: N, lignin: P, and (lignin+polyphenol): N, which collectively govern decomposition rates and nutrient release dynamics (Kaleeem Abbasi et al. 2015; Adhikari et al. 2024).

Crop residues exhibit substantial variability among species, cultivars within the same species, and in the relative biochemical composition of different plant components, which together determine the overall quality of whole-plant residues incorporated into soil (Redin et al. 2014; Coulibaly et al. 2020). Under field conditions, residues are typically incorporated as whole-plant biomass, and their decomposition behaviour reflects the integrated effects of these biochemical traits. However, existing findings remain inconsistent, with some studies reporting increased soil organic carbon (SOC) and improved nutrient availability following residue incorporation (Zhang et al. 2016; Liu et al. 2022), while others report nitrogen immobilization and reduced nutrient release (Meng et al. 2017; Kaleem Abbasi et al. 2015). These contrasting outcomes highlight the need for a deeper understanding of how cultivar-level biochemical differences in whole-plant residues influence decomposition processes, CO_2 emissions, and nutrient cycling. Given the widespread cultivation of maize and sorghum and the large quantities of residues they generate, understanding how cultivar-specific traits regulate residue behaviour in soil remains an important research priority.

To increase residue stability and prolong their benefits in soil, converting biomass into biochar through pyrolysis has received increasing attention. Biochar is a carbon-rich material produced under limited oxygen conditions and is widely reported to improve soil pH, cation exchange capacity, water-holding capacity, and microbial activity (Askeland et al. 2019; Purakayastha et al. 2019; Nguyen et al. 2020). In addition to these agronomic benefits, biochar has been proposed as a climate change mitigation strategy due to its potential to stabilize carbon in soil and alter greenhouse gas emissions (Lehmann 2007). Importantly, biochar properties vary strongly with feedstock type and pyrolysis temperature, which in turn regulate the balance between labile and recalcitrant carbon fractions and influence microbial activity, nutrient availability, and carbon mineralization processes (Biederman and Harpole 2013). Consequently, the effects of biochar application on soil CO_2 emissions remain highly variable across studies, with some reporting no statistically significant differences relative to unamended controls (Liu et al. 2016; He et al. 2016), others observing statistically significant increases in CO_2 emissions following biochar application (Thomazini et al. 2015; Wang et al. 2022), and other studies reporting statistically significant reductions in CO_2 emissions, often attributed to enhanced carbon stabilization and altered microbial activity (Zhao et al. 2020; Xiaoping et al. 2019; Wang et al. 2020; Duan et al. 2020).

Recent breeding efforts have developed maize and sorghum cultivars with improved stress tolerance and yield potential (Badu-Apraku et al. 2023; Kaliamoorthy et al. 2024). However, limited research has examined how these cultivar differences influence the biochemical quality of residues and the resulting effects on soil carbon and nutrient dynamics, particularly when residues are converted to biochar. Evidence from an earlier study on wheat residues showed significant cultivar-dependent differences in C, N, and P mineralization (Mbava et al. 2025), but that work focused only on raw feedstocks and did not include biochar. Thus, the influence of cultivar-specific biochemical traits on both feedstock and biochar-mediated soil processes remains an important and unresolved gap.

Maize and sorghum are primary staple crops for millions of people and serve as key components of cereal-based farming systems in sub-Saharan Africa (McMillen et al. 2022; Prasad et al. 2021). Their residues are abundant, yet generally underutilized, and represent a valuable resource for improving soil fertility and carbon storage. Understanding how cultivar-specific feedstocks and biochars influence CO_2 emissions and nutrient cycling is essential for developing residue management strategies that enhance soil health while contributing to climate resilience.

Therefore, this study investigated the effects of feedstock and biochar amendments produced from different maize and sorghum cultivars at two pyrolysis temperatures on soil CO_2-C emissions, soil pH, and the mineralization of nitrogen (NH_4^+-N and NO_3^--N) and phosphorus. It was hypothesised that (i) cultivar-specific differences in feedstock biochemical composition would result in contrasting CO_2-C emission patterns and nutrient mineralization responses, and (ii) biochars produced at higher pyrolysis temperatures would reduce CO_2-C emissions while enhancing soil pH and moderating nitrogen and phosphorus mineralization compared with feedstock amendments and lower-temperature biochars.

2 Materials and Methods

2.1 Biomass Collection and Feedstock Preparation

Two maize cultivars (R201 and SC701) and two sorghum cultivars (AS8 and PAN8816) were grown at Ukulinga

Research Farm, University of KwaZulu-Natal (29.667°S, 30.406°E; 811 m a.s.l.). The site has a long-term mean temperature of 18 °C and annual rainfall of 738 mm. The soil is classified as Westleigh (Soil Classification Working Group 1991), equivalent to a Luvisol (IUSS Working Group WRB, 2014), with a loamy texture (40.0% sand, 34.9% silt, and 24.4% clay), total C (1.9%), total N (0.17%), available P (8.87 mg kg^{-1}), exchangeable Ca (1043 mg kg^{-1}), Mg (313.7 mg kg^{-1}), K (91.94 mg kg^{-1}), and pH_{KCl} 4.73.

Each cultivar was planted on a 436 m^2 plot during the 2020–2021 season and fertilized with NPK (2:3:4) at recommended rates. At maturity, above-ground biomass was harvested, chopped, oven-dried, and milled to < 2 mm prior to pyrolysis to ensure homogeneity of the feedstock and uniform incorporation of both feedstock and biochar during the laboratory incubation experiment. Milling was conducted before pyrolysis to minimize post-pyrolysis fragmentation and material loss associated with the brittle, powdery nature of biochar. This pre-processing step was adopted for experimental control under laboratory conditions and is not intended to represent a practical residue management approach at farm scale.

2.2 Biochar Production

Feedstocks were pyrolyzed at 350 °C and 650 °C in a laboratory muffle furnace under oxygen-limited conditions using ceramic crucibles sealed with fitted lids, following a slow-pyrolysis protocol (heating rate of 10 °C min^{-1}; residence time of 2 h) as described by Vilakazi et al. (2023). Oxygen limitation was achieved by sealing the crucibles to restrict air ingress during heating. After cooling to room temperature, biochars were weighed and stored in airtight containers. Treatments were labelled according to cultivar and pyrolysis temperature (e.g., R201-B350, AS8-B650), with a full description provided in Table 1.

2.3 Feedstock and Biochar Characterization

Initial chemical and physical properties of feedstocks and biochars were determined before incubation (Table 2). Proximate analysis (moisture, volatile matter, ash, fixed C) followed ASTM 1762-84 procedures (Vilakazi et al. 2023). Total C and N were measured using a LECO TruMac CNS analyzer, and total H using a LECO CHS analyzer (Leco Corporation 2012). Total O was calculated by difference as described by Vilakazi et al. (2023). Feedstock and biochar pH were measured in 1 M KCl (1:10 w/v) following Enders et al. (2012). Total P was quantified after sulphuric acid-hydrogen peroxide digestion and colorimetric determination using the ascorbic acid method (Okalebo et al. 2002). Lignin content (feedstock only) was determined using the Van Soest detergent method (Van Soest et al. 1991). Exchangeable bases and CEC were measured using 1 M ammonium acetate at pH 7 (Okalebo et al. 2002; Anderson and Ingram 1994). Surface functional groups were identified using FTIR spectroscopy following Vilakazi et al. (2023), using spectra in the 4000–400 cm^{-1} range and band assignments according to Coates (2000). Surface morphology was examined using a Carl Zeiss EVO LS15 SEM; samples were gold-coated using a Quorum Q150R ES sputter coater prior to imaging.

Table 1 Description of different treatments used in the study

Treatment	Description
AS8-R	Sorghum experimental hybrid cultivar AS8 feedstock/residues
PAN8816-R	Sorghum commercial cultivar PAN8816 feedstock/residues
R201-R	Maize commercial cultivar R201 feedstock/residues
SC701-R	Maize commercial cultivar SC701 feedstock/residues
AS8-B350	Sorghum experimental hybrid cultivar AS8 biochar produced 350 °C
PAN8816-B350	Sorghum commercial cultivar PAN8816 biochar produced 350 °C
R201-B350	Maize commercial cultivar R201 biochar produced 350 °C
SC701-B350	Maize commercial cultivar SC701 biochar produced 350 °C
AS8-B650	Sorghum experimental hybrid cultivar AS8 biochar produced 650 °C
PAN8816-B650	Sorghum commercial cultivar PAN8816 biochar produced 650 °C
R201-B650	Maize commercial cultivar R201 biochar produced 650 °C
SC701-B650	Maize commercial cultivar SC701 biochar produced 650 °C

Treatment codes consist of the cultivar identifier (AS8, PAN8816, R201, SC701), followed by the amendment type (R = residue/feedstock; B = biochar) and, where applicable, the pyrolysis temperature (350–650 °C)

2.4 Soil Collection and Preparation

Soil for the incubation was collected from the same site before planting biomass (0–20 cm depth), air-dried, and sieved to < 2 mm. A subsample was analysed for baseline properties (texture, total C and N, available P, exchangeable cations, and pH).

2.5 Incubation Experiment for CO_2 Emissions

Soil (100 g dry-weight equivalent) was amended with feedstock or biochar at a rate equivalent to 10 t C ha^{-1}. This rate was calculated assuming a soil depth of 0–20 cm and a bulk density of 1.24 g cm^{-3}, corresponding to a soil mass of 2.48 × 106 kg ha^{-1}. Accordingly, an application rate

Table 2 Biochemical composition of feedstock and biochars of different cultivars of maize and sorghum

Crop type	Treatment	Total C	Total N	Total P	Fixed C	Ash	Lignin	pH	C:N	C:P
					%					
Maize	R201-R	38.89^{a}	0.32^{a}	0.47^{c}	17.24^{c}	6.50^{a}	38.57^{d}	5.09^{b}	153.30^{b}	82.76^{abc}
	SC701-R	33.29^{a}	1.07^{c}	0.38^{b}	12.23^{a}	16.73^{d}	4.18^{a}	6.75^{d}	31.08^{a}	88.42^{abcd}
	R201B-350	55.37^{bcd}	0.97^{c}	0.84^{h}	38.03^{b}	11.44^{b}		9.46^{f}	57.02^{a}	66.19^{a}
	SC701-B350	50.30^{bc}	1.57^{d}	0.73^{g}	32.59^{e}	28.59^{h}		9.75^{h}	32.11^{a}	69.23^{a}
	R201-B650	60.66^{d}	0.66^{b}	0.82^{h}	47.13^{h}	18.34^{e}		10.62^{k}	92.25^{ab}	74.00^{ab}
	SC701-B650	57.78^{cd}	0.99^{c}	0.69^{f}	26.59^{d}	46.13^{k}		11.4^{l}	58.42^{a}	83.74^{abc}
Sorghum	AS8-R	33.09^{a}	0.48^{a}	0.34^{a}	14.10^{b}	11.48^{b}	24.79^{c}	4.26^{a}	68.87^{a}	96.34^{bcd}
	PAN8816-R	37.62^{a}	1.46^{d}	0.33^{a}	13.17^{ab}	14.32^{c}	10.21^{b}	5.82^{c}	25.78^{a}	113.05^{d}
	AS8-B350	52.51^{bcd}	1.11^{c}	0.62^{e}	36.47^{f}	20.19^{f}		9.57^{g}	47.49^{a}	84.69^{abc}
	PAN8816-B350	49.74^{bc}	2.84^{e}	0.62^{e}	32.81^{e}	26.88^{g}		9.09^{e}	17.50^{a}	79.81^{ab}
	AS8-B650	59.59^{d}	0.69^{b}	0.56^{d}	32.48^{e}	36.00^{i}		10.58^{j}	86.00^{ab}	107.06^{cd}
	PAN8816-B650	49.33^{d}	1.61^{d}	0.54^{d}	27.51^{d}	41.82^{j}		10.44^{i}	30.72^{a}	91.37^{abcd}
	SE	1.64	0.033	5.31	0.223	0.176	0.545	4.81	15.92	4.91

Total C = total carbon content of the feedstock or biochar; Total N = total nitrogen content of the feedstock or biochar; Total P = total phosphorus content of the feedstock or biochar; Fixed C = fixed carbon content of the feedstock or biochar; Ash = ash content of the feedstock or biochar; Lignin = lignin content of the feedstock; pH = pH of the feedstock or biochar; C:N = carbon to nitrogen ratio of the feedstock or biochar; C:P = carbon to phosphorus ratio of the feedstock or biochar. Values are means. The standard error (SE) is shown for each variable. Different lowercase letters within the same column indicate significant differences among treatments according to the Tukey–Kramer multiple comparison test at $p< 0.05$

NB: The feedstock and biochar properties presented in Table 2 are provided to characterize the initial material composition used in the incubation experiment and to contextualize subsequent CO_2emissions and nutrient mineralization responses, recognizing that variations in elemental ratios, pH, and nutrient concentrations reflect inherent residue properties and concentration effects associated with pyrolysis rather than net nutrient gains

of 10 t C ha^{-1} is equivalent to 0.403 g C per 100 g soil. The mass of feedstock or biochar added to each incubation unit was adjusted based on the total carbon concentration of each material to ensure equivalent carbon inputs across treatments. An unamended soil served as the control. The experiment was arranged in a completely randomized design with 13 treatments (Table 1) and three replicates. Amended soils were thoroughly mixed and placed into 100 mL plastic containers, which were slowly wetted to 50% water-filled pore space. These containers were then placed inside 500 mL airtight plastic incubation jars to allow headspace for gas accumulation. A vial containing 25 mL of 1 M NaOH was placed inside each jar to trap CO_2 emitted during incubation. The jars were tightly sealed and incubated aerobically in the dark at a constant temperature of 25 °C for 120 days. Soil moisture was maintained at 50% water-filled pore space throughout the incubation period by regular gravimetric monitoring and adjustment based on weight loss. The NaOH traps were replaced at each sampling interval to prevent saturation and ensure continuous CO_2 capture. Sampling was conducted at 0, 3, 7, 14, 21, 28, 42, 56, 84, 112, and 120 days after incubation initiation. Trapped CO_2 was quantified by precipitating carbonates with $BaCl_2$, followed by back-titration of the residual NaOH with standardized 0.5 M HCl using phenolphthalein as an indicator (Franzluebbers et al. 2000). Cumulative CO_2-C was calculated by summing emissions across sampling intervals and expressed on a soil dry-weight basis.

2.6 Destructive Sampling for N and P Mineralization and Soil pH

A parallel incubation experiment was established for the determination of NH_4^+-N, NO_3^--N, extractable P, and soil pH using identical treatments to the CO_2 incubation. Destructive sampling was performed, with separate incubation containers prepared for each sampling date to avoid disturbance of remaining soil. For each treatment and sampling date, 100 g of soil (dry-weight equivalent) was thoroughly mixed with feedstock or biochar at a rate equivalent to 10 t C ha^{-1} and placed into 500 mL plastic containers. The containers were drilled 4 holes near the top rim to facilitate gas exchange and incubated aerobically in the dark at a constant temperature of 25 °C for 120 days. Soils were slowly wetted to field capacity and maintained at this moisture level throughout the incubation period by regular gravimetric adjustment based on weight loss.

Destructive sampling was conducted at 0, 7, 14, 21, 28, 42, 56, 84, 112, and 120 days of incubation. At each sampling time, the entire soil contents of each container were removed and homogenized prior to analysis. Mineral nitrogen (NH_4^+-N and NO_3^--N) was extracted using 2 M KCl and quantified calorimetrically using a Thermo Scientific Gallery Discrete Autoanalyser (Rayment and Lyons 2011). Extractable phosphorus was determined using the AMBIC-2 extraction method (Non-Affiliated Soil Analysis Work and Soil Science Society of South Africa, 1990). Soil pH was measured in 1 M KCl at a soil-to-solution ratio of 1:2.5 using a calibrated pH meter.

2.7 Statistical Analysis

Data were analysed using a two-way analysis of variance (ANOVA) in GenStat (20th Edition; Payne et al. 2011) to test the main effects of cultivar and amendment type, as well as their interaction, on CO_2-C emissions, NH_4^+-N, NO_3^--N, extractable P, and soil pH. Where significant effects were detected ($p<0.05$), treatment means were compared using Tukey-Kramer multiple comparison tests. Statistical significance is indicated in tables and figures using lowercase letters. Pearson's correlation analysis was conducted using OriginPro 2024 (OriginLab Corporation, Northampton, USA) to examine relationships among cumulative CO_2-C emissions, mineral N, extractable P, soil pH, and selected biochemical properties of the amendments.

3 Results

3.1 CO_2-C Emissions as Influenced by Feedstock and Biochar from Maize and Sorghum Cultivars

CO_2-C emissions differed significantly among treatments throughout the incubation period ($p<0.001$). Cultivar-specific differences and pyrolysis temperature exerted stronger effects on CO_2-C emissions than crop type. Emissions increased rapidly from day 0, peaking between 21 and 42 days across all treatments before declining thereafter (Fig. 1). The control consistently showed the lowest emissions early in the incubation but exceeded all biochar treatments after day 84. Feedstocks produced substantially higher CO_2-C emissions than their corresponding biochars for both crops (Table 3). Among maize cultivars, SC701 feedstock recorded the highest CO_2-C emission (308 mg CO_2-C kg^{-1} soil), an 89% increase relative to the control (162.8 mg CO_2-C kg^{-1} soil). Emissions from R201 treatments did not differ significantly across feedstock and biochar. For sorghum, PAN8816 feedstock exhibited the highest CO_2-C emissions (496.8 mg CO_2-C kg^{-1} soil), representing a 205% increase compared to the control. Biochar emissions generally decreased with increasing pyrolysis temperature, with B650 consistently producing the lowest CO_2-C among most cultivars.

3.2 Cumulative CO_2-C Emissions

Cumulative CO_2-C emissions varied significantly among treatments ($p<0.001$), steadily increasing over the 120 days of incubation (Fig. 2). The control showed the lowest cumulative CO_2-C throughout. For all cultivars except R201, cumulative emissions followed the order; feedstock > B350 > B650. Among maize-derived treatments,

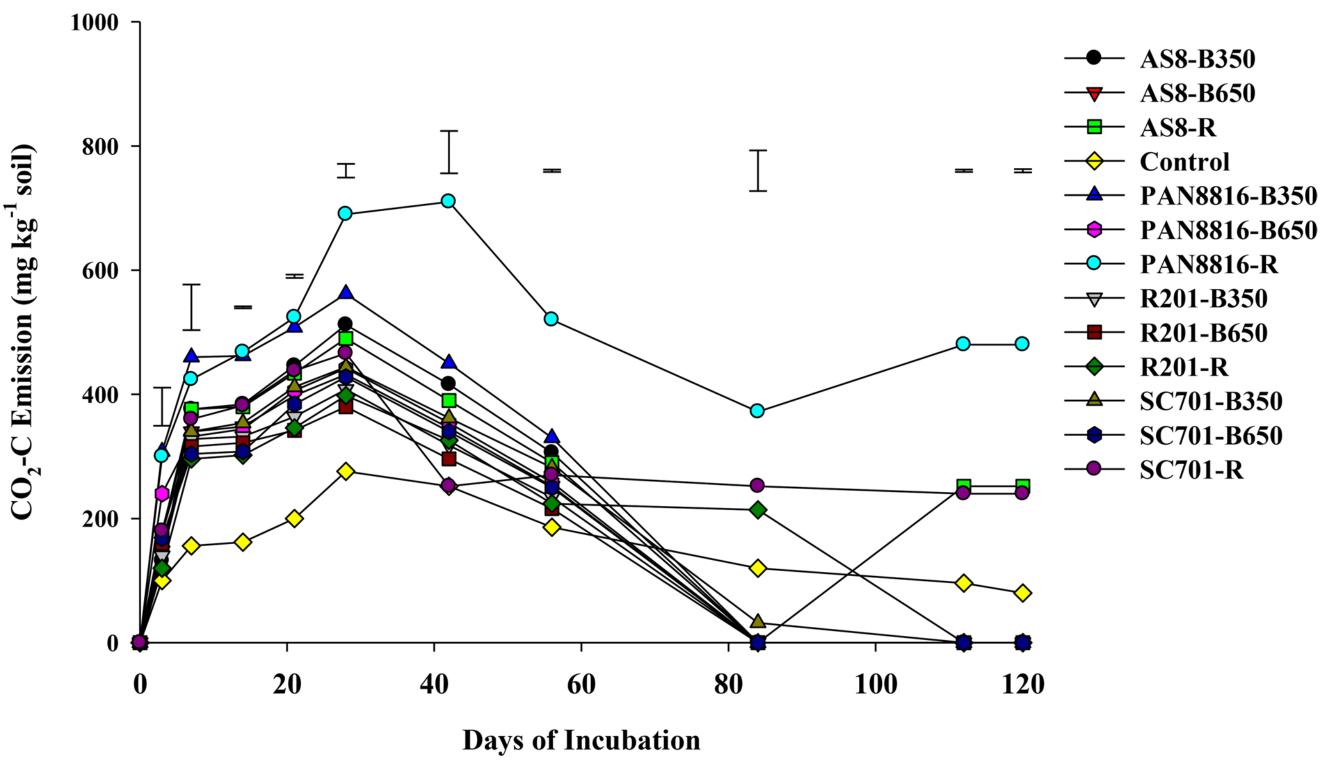

Fig. 1 Trends in CO_2-C emissions during the 120-day incubation period for soils amended with feedstock (R) and biochar pyrolyzed at 350 °C (B350) and at 650 °C (B650) from different maize (R201 & SC701) and sorghum (AS8 & PAN8816) cultivars along with the control treatment. Vertical error bars represent the standard error (±SE) of treatment means at each sampling time

Table 3 Average CO_2-C emissions, cumulative CO_2-C (mg CO_2-C kg^{-1} soil), nitrate-N (mg N kg^{-1} soil), ammonium-N (mg N kg^{-1} soil) pH and extractable phosphorus (mg P kg^{-1} soil) in soils amended with feedstock and biochar of different cultivars of maize and sorghum during a 120 days incubation

Crop type	Treatment	CO_2-C Emission	Cum. CO_2-C Emission	NH_4^+-*N*	NO_3^--*N*	Extractable *P*	pH
			mg kg^{-1} soil				
Maize	**R201-R**	222.60^{bc}	1426.00^{b}	6.72^{a}	8.10^{a}	7.34^{e}	4.99^{b}
	SC701-R	308.00^{e}	1777.00^{g}	12.57^{c}	11.35^{bc}	6.58^{d}	5.03^{bc}
	R201-B350	212.40^{bc}	1453.00^{bc}	7.87^{a}	10.23^{b}	10.10^{g}	5.24^{d}
	SC701-B350	240.20^{cd}	1623.00^{de}	12.66^{c}	13.40^{d}	8.08^{f}	5.26^{d}
	R201-B650	203.20^{b}	1404.00^{b}	7.20^{a}	13.78^{d}	10.96^{h}	5.40^{f}
	SC701-B650	218.20^{bc}	1484.00^{bc}	11.37^{bc}	19.29^{f}	7.99^{f}	5.36^{ef}
Sorghum	**AS8-R**	304.00^{e}	1803.00^{g}	12.02^{bc}	12.40^{cd}	6.11^{c}	5.12^{c}
	PAN8816-R	496.80^{f}	2655.00^{i}	16.82^{e}	24.92^{g}	5.07^{b}	4.82^{a}
	AS8-B350	257.20^{d}	1727.00^{fg}	12.36^{c}	16.40^{e}	7.46^{e}	5.29^{de}
	PAN8816-B350	308.00^{e}	2141.00^{h}	17.14^{e}	27.00^{h}	6.82^{d}	5.36^{ef}
	AS8-B650	226.40^{bcd}	1544.00^{cd}	10.52^{b}	17.37^{e}	7.36^{e}	5.43^{f}
	PAN8816-B650	237.60^{cd}	1644.00^{ef}	12.04^{bc}	18.91^{f}	6.79^{d}	5.43^{f}
	Control	162.80^{a}	939.00^{a}	15.17^{d}	16.44^{e}	2.67^{a}	4.84^{a}
	SE	6.83	20.51	0.33	0.32	0.07	0.02

Different lowercase letters within the same column indicate significant differences among treatments ($p<0.05$)

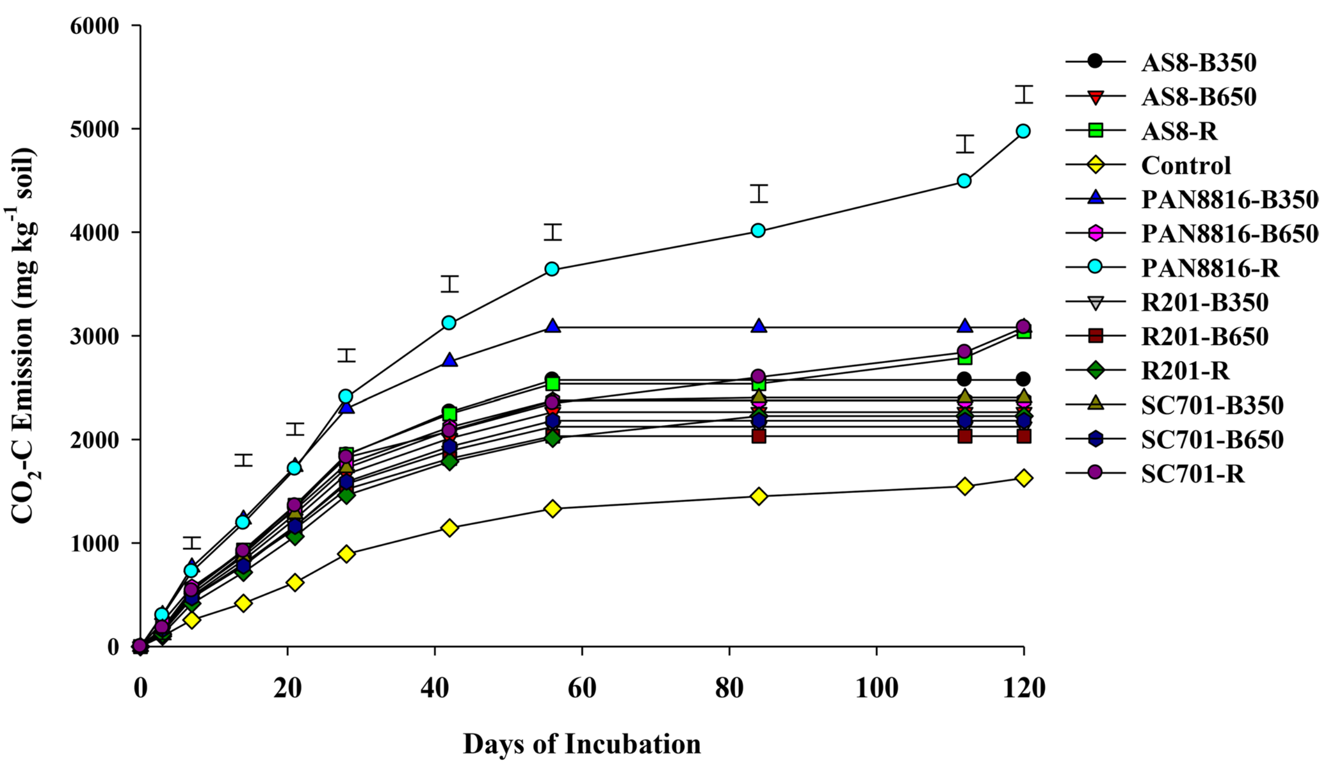

Fig. 2 Trends in cumulative CO_2-C emissions during the 120-day incubation period for soils amended with feedstock (R) and biochar pyrolyzed at 350 °C (B350) and at 650 °C (B650) from different maize (R201 & SC701) and sorghum (AS8 & PAN8816) cultivars along with the control treatment. Vertical error bars represent the standard error (± SE) of treatment means at each sampling time

SC701 feedstock produced the highest cumulative CO_2-C emission (1777 mg CO_2-C kg^{-1} soil). The SC701-B350 biochar also exhibited significantly higher cumulative CO_2-C emissions (1623 mg CO_2-C kg^{-1} soil) than the remaining maize treatments, which did not differ significantly from one another. For sorghum, PAN8816 feedstock recorded the highest cumulative emissions (2655.4 mg CO_2-C kg^{-1} soil), and AS8-B650 recorded the lowest (1544 mg CO_2-C kg^{-1}

soil) (Table 3). These trends confirm that cumulative CO_2-C emissions were strongly influenced by cultivar and pyrolysis temperature.

3.3 Effects of Feedstock and Biochar from Different Cultivars of Maize and Sorghum on Ammonium-N Mineralization

Ammonium-N (NH_4^+-N) concentrations differed significantly among treatments during the incubation period ($p<0.001$). Across all treatments, NH_4^+-N increased rapidly from the start of the incubation, peaked within 14 days, and then declined sharply thereafter (Fig. 3). R201 feedstock and its biochars consistently recorded the lowest NH_4^+-N concentrations during the first 21 days compared to other treatments. Ammonium-N was influenced more by cultivar-specific differences and pyrolysis temperature than by crop type (Table 3). Among maize cultivars, SC701 treatments generally showed higher NH_4^+-N concentrations than R201, although NH_4^+-N for both cultivars remained lower than the control (15.17 mg N kg^{-1} soil). For sorghum, PAN8816 feedstock (16.82 mg N kg^{-1} soil) and PAN8816-B350 (17.14 mg N kg^{-1} soil) had the highest NH_4^+-N concentrations, whereas all AS8 treatments were lower than the control (Table 3).

3.4 Nitrate-N (NO_3^--N) Mineralization

Nitrate-N concentrations differed significantly among treatments during incubation ($p<0.001$). Cultivar-specific differences and pyrolysis temperature strongly influenced NO_3^--N mineralization (Fig. 4). NO_3^--N increased slowly during the first 21 days, followed by a steady rise until day 84, after which concentrations stabilised for most treatments (Fig. 4). Among maize cultivars, R201 feedstock recorded the lowest NO_3^--N release (8.1 mg N kg^{-1} soil), which was 103% lower than the control (16.44 mg N kg^{-1} soil). The highest maize NO_3^--N concentration occurred in SC701-B650 (19.29 mg N kg^{-1} soil). Among sorghum cultivars, AS8 feedstock had the lowest NO_3^--N (12.4 mg N kg^{-1} soil), while PAN8816-B350 recorded the highest (27 mg N kg^{-1} soil), a 64% increase relative to the control (Table 3).

3.5 Extractable P Mineralization

Extractable P differed significantly across all treatments and sampling days ($p<0.001$). Extractable-P concentration increased rapidly from day 0, reaching peak values around day 28 for most treatments (Fig. 5). The control consistently released the lowest extractable P. Across all

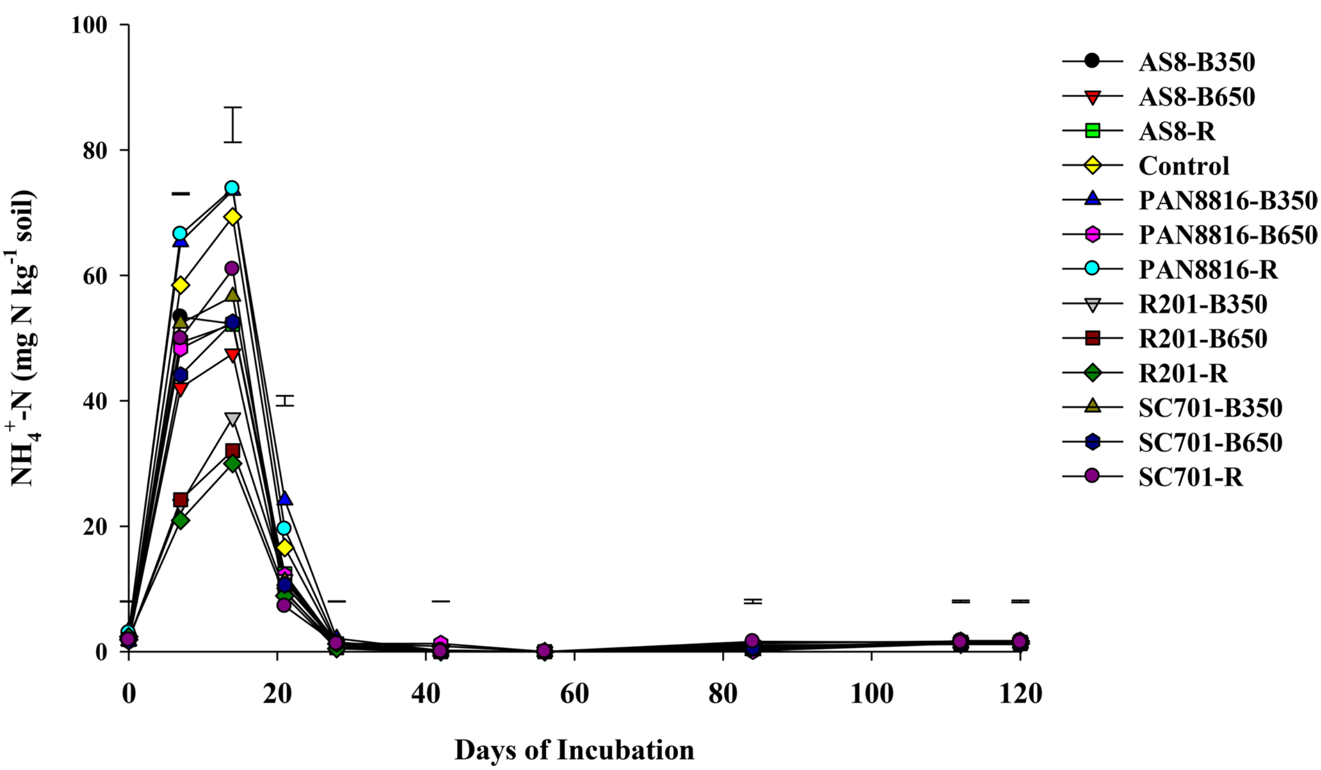

Fig. 3 Trends in NH_4^+-N release during the 120-day incubation period for soils amended with feedstock (R) and biochar pyrolyzed at 350 °C (B350) and at 650 °C (B650) from different maize (R201 & SC701) and sorghum (AS8 & PAN8816) cultivars along with the control treatment. Vertical error bars represent the standard error (±SE) of treatment means at each sampling time

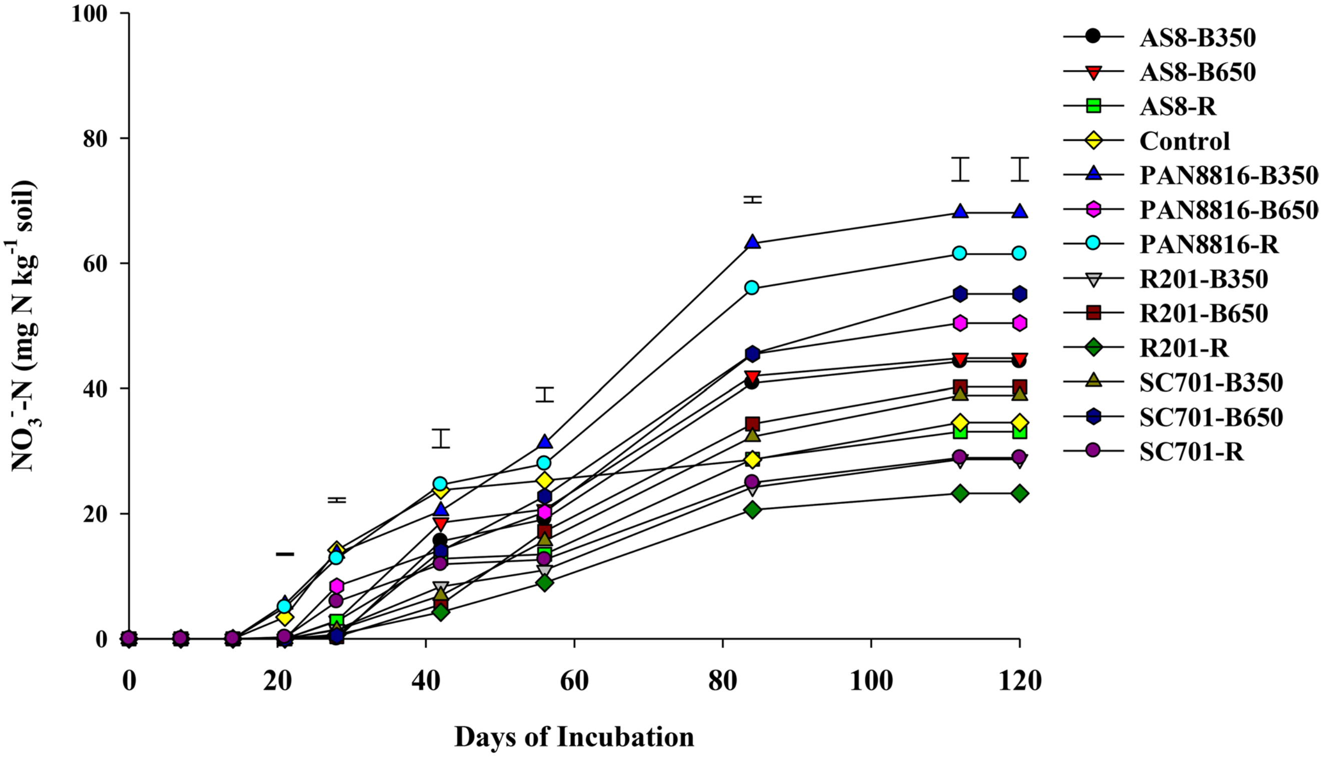

Fig. 4 Trends in NO_3^--N release during the 120-day incubation period for soils amended with feedstock (R) and biochar pyrolyzed at 350 °C (B350) and at 650 °C (B650) from different maize (R201 & SC701) and sorghum (AS8 & PAN8816) cultivars along with the control treatment. Vertical error bars represent the standard error (±SE) of treatment means at each sampling time

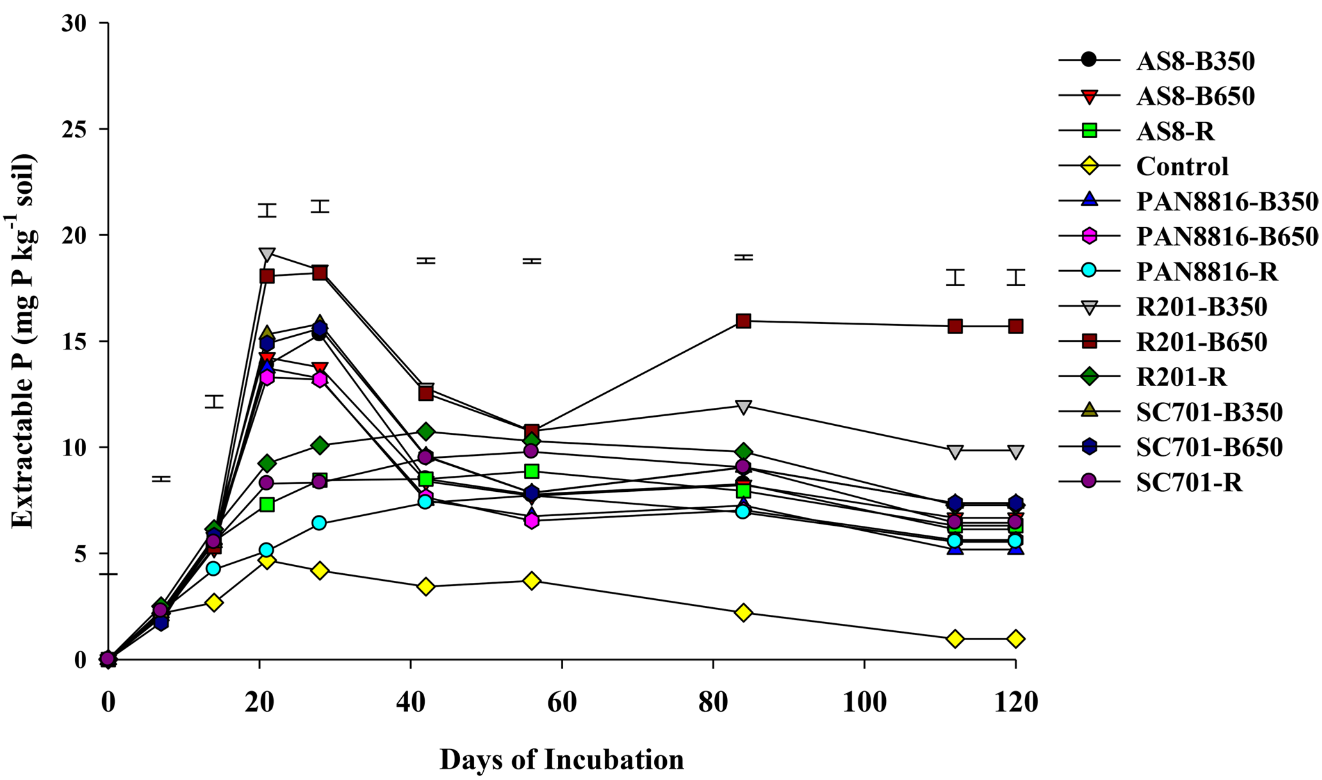

Fig. 5 Trends in extractable P release during the 120-day incubation period for soils amended with feedstock (R) and biochar pyrolyzed at 350 °C (B350) and at 650 °C (B650) from different maize (R201 & SC701) and sorghum (AS8 & PAN8816) cultivars along with the control treatment. Vertical error bars represent the standard error (±SE) of treatment means at each sampling time

cultivars, feedstocks produced lower extractable P than their respective biochars across all cultivars. Among maize cultivars, SC701 feedstock produced the lowest extractable P (6.58 mg P kg^{-1} soil), whereas R201-B650 had the highest (10.96 mg P kg^{-1} soil). For sorghum, PAN8816 feedstock recorded the lowest extractable P (5.07 mg P kg^{-1} soil), while AS8-B350 (7.46 mg P kg^{-1} soil) and AS8-B650 (7.36 mg P kg^{-1} soil) produced the highest concentrations. Differences in extractable P were primarily driven by cultivar, while the effect of pyrolysis temperature was cultivar-specific and significant only for R201-derived biochars (Table 3).

3.6 Soil pH

Soil pH differed significantly among treatments during incubation ($p<0.001$), increasing from day 0 and stabilizing after day 84 (Fig. 6). The lowest pH was recorded in the control (4.84), which did not differ significantly from PAN8816 feedstock (4.82). Across all cultivars, soil pH generally increased in the order; feedstock < B350 < B650. Biochars, especially B650, markedly increased soil pH relative to feedstocks, with cultivar-specific differences also contributing to pH variability (Table 3).

3.7 Relationships among CO_2-C Emissions, Nutrient Mineralization, Soil pH, and Residue Biochemical Properties

Pearson correlations (Table 4) showed significant negative relationships between CO_2-C emissions and residue total P (r= −0.67), total C (r= −0.60), fixed C (r= −0.60), soil pH (r= −0.71), and extractable P (r= −0.72). CO_2-C emissions were positively correlated with C: P ratio (r=0.64), NH_4^+-N (r=0.75), and cumulative CO_2-C (r=0.96). Cumulative CO_2-C was positively correlated with NO_3^--N (r=0.70) and NH_4^+-N (r=0.86) and negatively correlated with residue total P (r= −0.56). Extractable P was positively correlated with residue total P (r=0.89), total C (r=0.70), and fixed C (r=0.81), and negatively correlated with C: P ratio (r= −0.75) and NH_4^+-N (r= −0.71).

NH_4^+-N showed positive correlations with residue total N (r=0.76) and NO_3^--N (r=0.81), and negative correlation with C: N ratio (r= −0.80). NO_3^--N correlated positively with residue total N (r=0.75) and negatively with C: N ratio (r= −0.60). Soil pH correlated positively with residue total P (r=0.67), total C (r=0.82), fixed C (r=0.77), and residue pH (r=0.85). Additional correlations among residue properties are presented in Table 4.

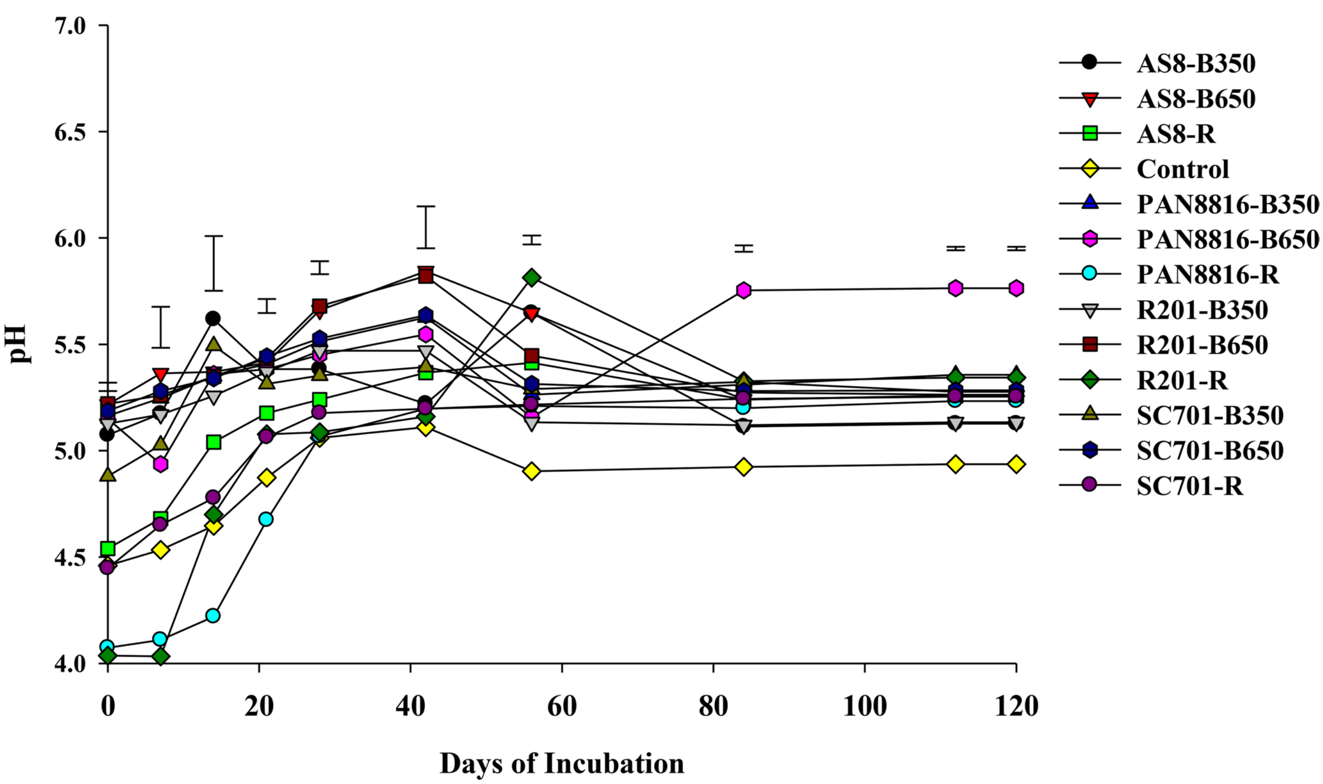

Fig. 6 Trends in soil pH during the 120-day incubation period in soils amended with feedstock (R) and biochar pyrolyzed at 350 °C (B350) and at 650 °C (B650) from different maize (R201 & SC701) and sorghum (AS8 & PAN8816) cultivars along with the control treatment. Vertical error bars represent the standard error (±SE) of treatment means at each sampling time

Table 4 Pearson's correlation coefficients (r) describing relationships between CO_2-C emissions, cumulative CO_2-C, mineral nitrogen forms (NH_4^+-N and NO_3^--N), extractable phosphorus, soil pH, and the biochemical properties of feedstock and biochars derived from different maize and sorghum cultivars

Parameter	CO_2-C Emission	Cumulative CO_2-C	Extractable *P*	NH_4^+-*N*	NO_3^--*N*	pH	Residue Ash	Residue C: *N*	Residue C: *P*	Residue Fixed C	Residue TN	Residue pH	Residue TC
Cumulative CO_2-C	0.96*												
Extractable P	−0.72*	−0.71*											
NH_4^+-N	0.75*	0.86*	−0.71*										
NO_3^--N	0.54	0.70*	−0.42	0.81*									
pH	−0.71*	−0.54	0.52	−0.19	0.14								
Residue Ash	−0.28	−0.16	−0.018	0.22	0.47	0.69*							
Residue C: N	−0.45	−0.57	0.32	−0.80*	−0.60*	−0.07	−0.34						
Residue C: P	0.64*	0.57	−0.75*	0.43	0.42	−0.35	0.09	−0.07					
Residue Fixed C	−0.6*	−0.46	0.81*	−0.33	0.03	0.77*	0.25	0.02	−0.57				
Residue TN	0.32	0.54	−0.25	0.76*	0.75*	0.20	0.33	−0.76*	−0.10	0.14			
Residue pH	−0.55	−0.41	0.56	−0.12	0.25	0.85*	0.75*	−0.22	−0.35	0.79*	0.26		
Residue TC	−0.60*	−0.48	0.70*	−0.31	0.15	0.82*	0.55	0.06	−0.36	0.89*	0.07	0.92*	
Residue TP	−0.67*	−0.56	0.89*	−0.42	−0.11	0.67*	0.26	0.04	−0.78*	0.91*	0.10	0.77*	0.85*

Residue TC = total carbon content of feedstock/biochar; Residue TP = total phosphorus content of feedstock/biochar; Residue TN = total nitrogen content of feedstock/biochar; Residue pH = pH of feedstock/biochar; Residue Ash = ash content of feedstock/biochar; Residue C: N = carbon-to-nitrogen ratio of feedstock/biochar; Residue C: P = carbon-to-phosphorus ratio of feedstock/biochar

Significant correlations at $p < 0.05$ are indicated by (*)

4 Discussion

The results of this incubation study demonstrate that CO_2-C emissions and nutrient mineralization were primarily governed by cultivar-specific biochemical properties rather than broad crop type. Within both maize and sorghum, one cultivar consistently exhibited higher CO_2-C emissions, NH_4^+-N, NO_3^--N, and extractable P, while the corresponding cultivar showed lower values for these variables. This pattern reflects differences in residue biochemical composition among cultivars, especially in total C, total N, total P, and their C: N and C: P ratios, which control microbial decomposition processes (Mbava et al. 2025; Ntonta et al. 2024). In this study, cultivars with higher nutrient concentrations and narrower C: N and C: P ratios (Table 2) promoted greater microbial activity, as evidenced by increased CO_2-C emissions and enhanced N and P mineralization. These relationships are supported by significant correlations between CO_2-C emissions and residue C: N and C: P ratios, as well as positive associations with residue total N and total P (Table 4), indicating strong stoichiometric control over decomposition dynamics. This pattern aligns with You et al. (2020), who attributed residue decomposition differences primarily to chemical composition. The present findings therefore challenge the assumption that crop type alone determines residue behavior in soil and instead show the importance of cultivar level biochemical traits when selecting residues for soil fertility enhancement and carbon management.

The influence of pyrolysis temperature on soil responses was not uniform across cultivars but instead depended strongly on cultivar-specific feedstock properties. Biochars derived from the maize cultivar R201 exhibited clear temperature dependent effects, particularly for extractable P whereas biochars produced from SC701 and the sorghum cultivars AS8 and PAN8816 showed comparatively smaller or non-significant differences between 350 °C and 650 °C. This indicates that the impact of pyrolysis temperature depends on the biochemical composition of the original feedstock, rather than temperature acting as an independent driver (Conz et al. 2017). Differences in total P concentration, ash content, fixed carbon, and elemental stoichiometry among cultivars (Table 2) determined how increasing pyrolysis temperature altered nutrient availability and carbon stability. Such feedstock-temperature interactions support the findings of Khan et al. (2024), who reported that biochar performance is strongly influenced by the biochemical properties of the original feedstock. Across treatments, the pronounced increase in CO_2-C emissions during the early incubation period reflects rapid microbial activation following amendment addition. This initial pulse is consistent with the stimulation of heterotrophic respiration driven by readily available carbon substrates, as previously reported by Dumale Jr et al. (2009). Feedstock amended soils produced higher early CO_2-C emissions than biochar amended soils, indicating a greater proportion of labile carbon that was readily decomposed by soil microorganisms (Watanabe and Sato 2015). On the other hand, biochar amended soils exhibited lower CO_2-C emissions, reflecting the greater structural stability and aromaticity of biochar carbon, which limits microbial accessibility (Case et al. 2014). These findings align with earlier studies showing that biochar reduces CO_2-C release relative to unpyrolyzed residues due to its reduced labile carbon fraction and increased carbon recalcitrance (Zavalloni et al. 2011; Zhao et al. 2020; Duan et al. 2020).

The relatively similar CO_2-C emissions observed across biochar treatments, irrespective of pyrolysis temperature or cultivar, can be attributed to the shared biochemical characteristics of the biochars produced in this study. Although biochars were produced from different cultivars and temperatures, the feedstocks had similar total carbon concentrations and were subjected to the same pyrolysis conditions. This may explain the relatively small differences observed in carbon mineralization and CO_2-C emissions among biochar treatments. This interpretation is supported by the significant negative correlations between CO_2-C emissions and residue total C, fixed C, and pH (Table 4), suggesting that greater carbon stability and alkalinity suppress microbial decomposition. Therefore, the higher total C, fixed C content, and pH of biochars compared to feedstocks (Table 2) likely contributed to reduced microbial respiration and lower CO_2-C emissions. In contrast, feedstocks generally contained lower total C and fixed C and exhibited lower pH, providing more labile and readily decomposable carbon substrates that stimulated microbial activity and enhanced CO_2-C emission. These patterns are consistent with the findings of Hailegnaw et al. (2019), who reported that biochar contains chemically stabilized carbon that is less accessible to soil microorganisms, resulting in slower organic matter decomposition and reduced CO_2 emissions. This indicates that the stability of biochar carbon helps reduce microbial respiration and supports long-term soil carbon retention.

Differences among cultivars and pyrolysis temperatures became more pronounced when examining cumulative CO_2-C (Fig. 2). Among the treatments, cumulative CO_2-C emissions were generally higher for feedstocks and for biochars produced at 350 °C. This can be explained by the higher proportion of labile carbon fractions and volatile compounds retained at lower pyrolysis temperatures, which remain accessible to microbial metabolism and promote sustained carbon mineralization (Whitman et al. 2012). In contrast, biochars produced at 650 °C consistently resulted in lower cumulative CO_2-C emissions, as higher pyrolysis temperatures produce more aromatic and chemically stable carbon structures that are less accessible to microbial degradation (Peng et al. 2011). These temperature-driven differences are further supported by the significant negative correlations between cumulative CO_2-C emissions and fixed C content as well as biochar pH (Table 4), indicating that increased carbon stabilization and alkalinity suppress long-term microbial respiration. Similar reductions in cumulative CO_2-C emissions with increasing pyrolysis temperature have been reported by Deng et al. (2020), who attributed this effect to the enhanced stability and persistence of high-temperature biochars in soil. Clear cultivar specific differences were observed within both maize and sorghum, with some cultivars consistently producing higher cumulative CO_2-C emissions than others (SC701 and PAN8816 feedstocks), while others exhibited lower emissions (R201 treatments and AS8-B650). These differences were closely associated with differences in feedstock biochemical composition and their response to pyrolysis temperature, particularly in terms of fixed C formation and nutrient stoichiometry (Table 2). Cultivars that generated biochars with higher fixed C and more stable carbon fractions showed reduced cumulative CO_2-C emissions, whereas those retaining more labile carbon supported faster mineralization. These findings show that both cultivar choice and higher pyrolysis temperature influence cumulative carbon losses, highlighting the need to consider feedstock characteristics and processing conditions when aiming to retain carbon in soil.

Nitrogen mineralization patterns were strongly influenced by cultivar and pyrolysis temperature, reflecting the interaction between residue biochemical composition and microbial processes. Previous research has documented both increases and decreases in mineral N following residue or biochar application, depending on the balance between mineralization, immobilization, adsorption and nitrification processes (Yao et al. 2012; Zheng et al. 2013). The rapid initial increase in NH_4^+-N across treatments indicates ammonification, whereby microbial decomposition of organic nitrogen compounds including proteins and amino acids in residues and native soil organic matter released NH_4^+-N (van Midden et al. 2024). Similar early NH_4^+-N peaks have been reported following amendment addition or soil rewetting (Liyanage et al. 2022; Marouani et al. 2021). The subsequent decline in NH_4^+-N is consistent with nitrification, during which autotrophic microorganisms oxidize NH_4^+-N to NO_3^--N under aerobic conditions (Baskaran et al. 2020). The aerobic incubation setup used in this study, including perforated containers and controlled moisture conditions, favoured oxygen diffusion and nitrifier activity. Furthermore, the increase in NH_4^+-N observed in the control reflects mineralization of native soil organic nitrogen and microbial biomass turnover stimulated by favourable incubation conditions (Liyanage et al. 2022; Marouani et al. 2021).

Although this general sequence was consistent, the magnitude of NH_4^+-N and NO_3^--N responses differed among cultivars and pyrolysis temperatures. In the present study, several treatments (R201, SC701, AS8 and PAN8816-B650) reduced NH_4^+-N compared to the control, whereas PAN8816 feedstock and PAN8816-B350 increased NH_4^+-N. These contrasting responses indicate that nitrogen dynamics were influenced more by cultivar-specific residue biochemical composition than by whether the amendment was a feedstock or a biochar. Nitrogen mineralization is strongly regulated by residue C: N ratio, total N content and labile N fractions (Zavalloni et al. 2011). Residues with wide C: N ratios tend to promote temporary N immobilization because microorganisms require additional nitrogen to decompose carbon-rich substrates (Mbava et al. 2025). This mechanism is supported by the significant negative correlations between residue C: N ratio and both NH_4^+-N and NO_3^--N (Table 4). Treatments such as R201-R, R201-B650 and AS8-B650, characterized by wide C: N ratios, exhibited reduced mineral N availability. In contrast PAN8816 amendments consistently produced higher NH_4^+-N and NO_3^--N concentrations, corresponding with their higher total N content. The significant positive correlations between residue total N and mineral N fractions (Table 4) confirm that nitrogen supply can offset immobilization effects driven by wide C: N ratios and low total N.

Nitrate-N patterns further reflected these interactions. The gradual increase in NO_3^--N following the decline in NH_4^+-N indicates active nitrification under aerobic conditions (Mbava et al.2025). Cultivar PAN8816-based amendments resulted in higher NO_3^--N concentrations. In contrast, treatments associated with stronger immobilization and lower NH_4^+-N supply, such as R201-based amendments, exhibited reduced NO_3^--N. Similar reductions in mineral N following residue or biochar application have been reported for wheat residues and wood biochar (Zavalloni et al. 2011), switchgrass residues and pecan shell biochar (Novak et al. 2010), and maize biochar produced at moderate temperatures (ODUGBENRO et al. 2018). Reductions in mineral N do not always persist, for example Hossain Bai et al. (2014) reported minimal short-term effects of pinewood biochar (550 °C) on mineral N, followed by later changes linked to decomposition and microbial turnover. Therefore, nitrogen availability in amended soils reflects a balance between microbial demand driven by carbon quality and the total nitrogen supplied by the amendment, shaped by cultivar-specific biochemical composition and pyrolysis-induced changes in nutrient availability.

Similar to mineral N, extractable P mineralization was strongly influenced by cultivar-specific differences, with the effect of pyrolysis temperature varying depending on the feedstock. Extractable P increased rapidly from the beginning of the incubation period, indicating the presence of readily mineralizable or weakly bound P fractions in both feedstocks and biochars. Previous studies have shown that the impact of residue and biochar amendments on soil P availability can be either positive or negative, depending on feedstock chemistry and pyrolysis conditions (Amendola et al. 2017; Li et al. 2017; Liu et al. 2017). In the present study, however, all amendments increased soil extractable P compared to the control, indicating improved P availability. Phosphorus release following amendment addition is closely linked to the initial P content of the organic material. Mafongoya et al. (2000) reported that residues with total P contents exceeding 0.2% generally promote P mineralization rather than immobilization. All feedstocks and biochars used in this study exceeded this threshold (Table 2), and residue total P showed a significant positive correlation with extractable P (Table 4), directly explaining the increased extractable P concentrations observed across amended treatments. The highest extractable P concentrations were observed in R201 and AS8 derived amendments, particularly in their biochar treatments for maize and sorghum, respectively. These responses correspond with the higher total P contents of these residues and indicate that cultivar-level differences in P concentration played a dominant role in regulating P mineralization.

The enhanced extractable P observed in biochar amended soils reflects changes in P availability rather than net P production. During pyrolysis, organic matter is volatilized, resulting in the concentration of inorganic P within the biochar ash fraction, which can become readily extractable upon soil incorporation. In addition, increases in soil pH associated with biochar application reduce P sorption to Fe and Al oxides, thereby increasing P availability in acidic soils (Glaser and Lehr 2019). These mechanisms account for the increased extractable P measured in biochar treatments and are consistent with previous studies reporting increased soil available P following the application of crop residue and woody biochars (Zhai et al. 2015; Gupta et al. 2024). These findings indicate that cultivar-specific P content, coupled with pyrolysis induced concentration and soil chemical modification, governs extractable P dynamics in amended soils.

Application of feedstocks and biochars generally increased soil pH compared to the control, with pyrolysis temperature exerting a stronger influence than cultivar or crop type. Soil pH was consistently lower under feedstock treatments than under biochar treatments, reflecting the acidic nature of unpyrolyzed residues and the release of organic acids during early decomposition (Jin-Hua et al. 2011). In contrast, biochar-amended soils exhibited significantly higher pH values due to the alkaline nature of biochars formed during pyrolysis, particularly as a result of increased ash content and the formation of basic mineral compounds (Rizwan et al. 2023). These findings are consistent with previous reports of biochar's liming effect arising from its ash fraction and alkaline surface functional groups (Chintala et al. 2014; Silva et al. 2017). The positive correlations between soil pH, residue ash content, and residue pH further confirm that amendment chemical properties governed pH responses. Pyrolysis concentrates mineral ash and enhances alkalinity, thereby increasing biochar's buffering capacity against soil acidification (Nelissen et al. 2012). This explains the higher soil pH observed in 650 °C biochars compared with 350 °C biochars for maize cultivars. However, for sorghum cultivars, differences in residue ash content and pH between 350 °C and 650 °C biochars did not result in significant differences in soil pH. This suggests a buffering saturation effect, whereby both sorghum biochars supplied sufficient alkalinity to neutralize soil acidity, limiting further pH increases despite compositional differences. Similar saturation responses have been reported when biochar additions exceed the soil's buffering requirement (Coumaravel et al. 2011; Shetty et al. 2020).

The initial increase of pH in the control following incubation reflects ammonification of native soil organic nitrogen, a process that consumes protons and can temporarily increase soil pH after rewetting (Jin-Hua et al. 2011). As incubation progressed, the subsequent decline in pH corresponds to nitrification, during which oxidation of NH_4^+-N to NO_3^--N releases H^+ ions and promotes soil acidification (Butterly et al. 2013). In contrast, amended soils exhibited relatively stable pH throughout incubation, indicating that feedstocks and particularly biochars buffered biologically induced pH fluctuations through the supply of base cations and alkaline ash components. A short-term anomaly was observed at day 0, where soils amended with R201 and PAN8816 feedstocks exhibited lower pH than the control despite their intrinsic residue pH being higher than the initial soil pH. This transient decrease likely resulted from the rapid release of organic acids, phenolic compounds, and dissolved CO_2 during residue wetting and early microbial activation, temporarily acidifying the soil immediately after amendment incorporation (Anderson et al. 2018). As decomposition progressed, these effects diminished and soil pH increased and stabilized, especially in biochar amended treatments due to their greater buffering capacity. These results demonstrate that while feedstocks can induce short-term acidification during decomposition, biochar amendments exert a stronger and more sustained liming effect. The relative stability of pH in biochar-amended soils show biochar's capacity to regulate soil chemical conditions and mitigate pH fluctuations during nutrient cycling.

5 Conclusion

This study demonstrates that biochemical differences among the selected maize and sorghum cultivars exerted stronger control over carbon dioxide emissions, nitrogen and phosphorus mineralization, and soil pH than crop type alone. Within maize, SC701 feedstock produced the highest cumulative carbon dioxide emissions, whereas R201 feedstock and its biochars resulted in consistently lower emissions. For sorghum, PAN8816 feedstock generated the highest cumulative emissions, while AS8 biochar produced at 650 °C exhibited the lowest. These patterns reflect cultivar level differences in carbon content, nitrogen content, elemental ratios, and ash composition that regulated microbial decomposition and nutrient transformations. Pyrolysis temperature further modified these responses. Biochars produced at 650 °C generally reduced cumulative carbon dioxide emissions compared with feedstocks and 350 °C biochars, reflecting more carbon stabilization and resistance to microbial degradation. Higher temperature biochars also exerted stronger liming effects,, due to increased ash concentration and alkalinity. In contrast, biochars produced at 350 °C retained more labile fractions and in some cases promoted greater mineral nitrogen availability.

Nitrogen dynamics were governed by the balance between residue carbon quality and total nitrogen supply. Treatments with wide C: N ratios, including R201 feedstock and certain high-temperature biochars, were associated with reduced mineral nitrogen concentrations, consistent with enhanced microbial immobilization. In contrast, PAN8816 feedstock and its 350 °C biochar, characterized by higher total nitrogen content, increased both ammonium and nitrate concentrations. Phosphorus availability was strongly linked to initial residue phosphorus content and ash concentration, with R201 biochar produced at 650 °C and AS8 biochars generating the highest extractable phosphorus levels. These results show that carbon stabilization, nitrogen transformations, phosphorus availability, and soil acidity were controlled by cultivar-specific biochemical composition and pyrolysis temperature. Importantly, the dominance of cultivar-level effects observed reflects the specific maize and sorghum cultivars evaluated under controlled incubation conditions and should not be generalized to all genotypes. Therefore, residue management outcomes depend on feedstock biochemical composition and thermal pyrolysis temperature rather than broad crop type alone. Future field research is needed to verify these outcomes under variable environmental conditions and assess long-term impacts on soil fertility and crop performance.

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Author Contribution All authors contributed to the study conception and design. Material preparation, data collection, and analysis were performed by Nozibusiso Mbava. The first draft of the manuscript was written by Nozibusiso Mbava and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

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Data Availability The datasets used or analyzed during the current study are available from the corresponding author upon reasonable request.

Declarations

Competing Interests The authors have no competing interests to declare that are relevant to the content of this article.

References

Adhikari AD, Shrestha P, Ghimire R, Liu Z, Pollock DA, Acharya P, Aryal DR (2024) Cover crop residue quality regulates litter decomposition dynamics and soil carbon mineralization kinetics in semi-arid cropping systems. Applied Soil Ecology. https://doi.org/10.1016/j.apsoil.2023.105160

Aher G, Cihacek LJ, Cooper K (2017) An evaluation of C and N on fresh and aged crop residue from mixed long-term no-till cropping systems. Journal of Plant Nutrition. https://doi.org/10.1080/01904167.2016.1201505

Amendola C, Montagnoli A, Terzaghi M, Trupiano D, Oliva F, Baronti S, Miglietta F, Chiatante D, Scippa GS (2017) Short-term effects of biochar on grapevine fine root dynamics and arbuscular mycorrhizae production. Agric Ecosyst Environ. https://doi.org/10.1016/j.agee.2017.01.025

Anderson JM, Ingram JS (1994) Tropical soil biology and fertility: a handbook of methods. Soil Sci.

Anderson CR, Peterson ME, Frampton RA, Bulman SR, Keenan S, Curtin D (2018) Rapid increases in soil pH solubilise organic matter, dramatically increase denitrification potential and strongly stimulate microorganisms from the Firmicutes phylum. PeerJ. https://doi.org/10.7717/peerj.6090

Askeland M, Clarke B, Paz-Ferreiro J (2019) Comparative characterization of biochars produced at three selected pyrolysis temperatures from common woody and herbaceous waste streams. PeerJ. https://doi.org/10.7717/peerj.6784

Badu-Apraku B, Fakorede MA, Nelimor C, Osuman AS, Bonkoungou TO, Muhyideen O, Akinwale RO (2023) Recent advances in breeding maize for drought, heat and combined heat and drought stress tolerance in sub-saharan Africa. CABI Reviews 2023. https://doi.org/10.1079/cabireviews.2023.0011

Baskaran V, Patil PK, Antony ML, Avunje S, Nagaraju VT, Ghate SD, Nathamuni S, Dineshkumar N, Alavandi SV, Vijayan KK (2020) Microbial community profiling of ammonia and nitrite oxidizing bacterial enrichments from brackishwater ecosystems for mitigating nitrogen species. Sci Rep. https://doi.org/10.1038/s41598-020-62183-9

Biederman LA, Harpole WS (2013) Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. GCB bioenerg. https://doi.org/10.1111/gcbb.12037

Butterly CR, Baldock JA, Tang C (2013) The contribution of crop residues to changes in soil pH under field conditions. Plant Soil. https://doi.org/10.1007/s11104-012-1422-1

Case SD, McNamara NP, Reay DS, Whitaker J (2014) Can biochar reduce soil greenhouse gas emissions from a Miscanthus bioenergy crop? GCB bioenergy. https://doi.org/10.1111/gcbb.12052

Chintala R, Mollinedo J, Schumacher TE, Malo DD, Julson JL (2014) Effect of biochar on chemical properties of acidic soil. Arch Agron Soil Sci. https://doi.org/10.1080/03650340.2013.789870

Coates J (2000) Interpretation of infrared spectra, a practical approach. EAC. https://doi.org/10.1002/9780470027318.a5606

Conz RF, Abbruzzini TF, de Andrade CA, Milori DM, Cerri CE (2017) Effect of pyrolysis temperature and feedstock type on agricultural properties and stability of biochars. Agric Sci. https://doi.org/10.4236/as.2017.89067

Coulibaly SS, Touré M, Kouamé AE, Kambou IC, Soro SY, Yéo KI, Koné S (2020) Incorporation of crop residues into soil: a practice to improve soil chemical properties. Agric Sci. https://doi.org/10.4236/as.2020.1112078

Coumaravel K, Santhi R, Kumar VS, Mansour MM (2011) Biochar–A promising soil additive-A review. Agric Review

Deng B, Shi Y, Zhang L, Fang H, Gao Y, Luo L, Feng W, Hu X, Wan S, Huang W, Guo X (2020) Effects of spent mushroom substrate-derived biochar on soil CO_2 and N_2O emissions depend on pyrolysis temperature. Chemosphere. https://doi.org/10.1016/j.chemosphere.2019.125608

Duan M, Wu F, Jia Z, Wang S, Cai Y, Chang SX (2020) Wheat straw and its biochar differently affect soil properties and field-based greenhouse gas emission in a Chernozemic soil. Biol Fertil Soils. https://doi.org/10.1007/s00374-020-01479-4

Dumale WA Jr, Miyazaki T, Nishimura T, Seki K (2009) CO_2 evolution and short-term carbon turnover in stable soil organic carbon from soils applied with fresh organic matter. Geophys Res Lett. https://doi.org/10.1029/2008GL036436

Enders A, Hanley K, Whitman T, Joseph S, Lehmann J (2012) Characterization of biochars to evaluate recalcitrance and agronomic performance. Bioresour Technol. https://doi.org/10.1016/j.biortech.2012.03.022

Franzluebbers AJ, Haney RL, Honeycutt CW, Schomberg HH, Hons FM (2000) Flush of carbon dioxide following rewetting of dried soil relates to active organic pools. Soil Sci Soc Am J DOI. https://doi.org/10.2136/sssaj2000.642613x

Glaser B, Lehr VI (2019) Biochar effects on phosphorus availability in agricultural soils: A meta-analysis. Sci Rep. https://doi.org/10.1038/s41598-019-45693-z

Gupta RK, Vashisht M, Naresh RK, Dhingra N, Sidhu MS, Singh PK, Rani N, Al-Ansari N, Alataway A, Dewidar AZ, Mattar MA (2024) Biochar influences nitrogen and phosphorus dynamics in two texturally different soils. Sci Rep. https://doi.org/10.1038/s41598-024-55527-2

Hailegnaw NS, Mercl F, Pračke K, Száková J, Tlustoš P (2019) High temperature-produced biochar can be efficient in nitrate loss prevention and carbon sequestration. Geoderma. https://doi.org/10.1016/j.geoderma.2018.11.006

He X, Du Z, Wang Y, Lu N, Zhang Q (2016) Sensitivity of soil respiration to soil temperature decreased under deep biochar amended soils in temperate croplands. Appl Soil Ecol. https://doi.org/10.1016/j.apsoil.2016.08.018

Heerink N (2005) Soil fertility decline and economic policy reform in Sub-Saharan Africa. Land Use Policy. https://doi.org/10.1016/j.landusepol.2003.08.002

Hosseini Bai S, Xu CY, Xu ZH, Blumfield TJ, Wallace HM, Walton DA, Randall BW, Van Zwieten L (2014) August. Wood base biochar alters inorganic N. In XXIX International Horticultural Congress on Horticulture: Sustaining Lives, Livelihoods and Landscapes (IHC2014): https://doi.org/10.17660/ActaHortic.2016.1109.24

Jin-Hua YUAN, Ren-Kou XU, Ning W, Jiu-Yu LI (2011) Amendment of acid soils with crop residues and biochars. Pedosphere. https://doi.org/10.1016/S1002-0160(11)60130-6

Kaleeem Abbasi M, Mahmood Tahir M, Sabir N, Khurshid M (2015) Impact of the addition of different plant residues on nitrogen mineralization–immobilization turnover and carbon content of a soil incubated under laboratory conditions. Solid Earth. https://doi.org/10.5194/se-6-197-2015

Kaliamoorthy S, Gaddameedi A, Murugesan T, Gorthy S, Sravani B, Neelam N, Jayakumar J, Choudhary S, Kholovà J, Govindaraj M (2024) Exploring genotypic diversity in sorghum breeding lines for water-saving traits to enhance drought adaptation during the post-rainy season. Crop Sci. https://doi.org/10.1002/csc2.21285

Khan S, Irshad S, Mehmood K, Hasnain Z, Nawaz M, Rais A, Gul S, Wahid MA, Hashem A, Abd_Allah EF, Ibrar D (2024) Biochar production and characteristics, its impacts on soil health, crop production, and yield enhancement: a review. Plants. https://doi.org/10.3390/plants13020166

Lal R (2005) World crop residues production and implications of its use as a biofuel. Environ Int. https://doi.org/10.1016/j.envint.2004.09.005

Leco Corporation (2012) Truspec CN Carbon/ Nitrogen Determinator Instructions Manual. LECO Corporation, St Joseph, USA

Lehmann J (2007) Bio-energy in the black. Front Ecol Environ. https://doi.org/10.1890/15409295(2007)5[381:BITB]2.0.CO;2

Li X, Rubæk GH, Müller-Stöver DS, Thomsen TP, Ahrenfeldt J, Sørensen P (2017) Plant availability of phosphorus in five gasification biochars. Front Sustain Food Syst. https://doi.org/10.3389/978-2-88963-219-0

Liu X, Zheng J, Zhang D, Cheng K, Zhou H, Zhang A, Pan G (2016) Biochar has no effect on soil respiration across Chinese agricultural soils. Science of The Total Environment. https://doi.org/10.1016/j.scitotenv.2016.02.179

Liu S, Meng J, Jiang L, Yang X, Lan Y, Cheng X, Chen W (2017) Rice husk biochar impacts soil phosphorous availability, phosphatase activities and bacterial community characteristics in three different soil types. Appl Soil Ecol. https://doi.org/10.1016/j.apsoil.2017.03.020

Liu S, Li J, Liang A, Duan Y, Chen H, Yu Z, Pan H (2022) Chemical composition of plant residues regulates soil organic carbon turnover in typical soils with contrasting textures in northeast China plain. Agron. https://doi.org/10.3390/agronomy12030747

Liyanage M, Hanafi MM, Sulaiman MF, Ismail R, Gunaratne G, Dharmakeerthi S, Rupasinghe G, Mayakaduwa A (2022) Consequences of nitrogen mineralization dynamics for soil health restoration of degraded tea-growing soil using organic amendments. Chil J Agric Res. https://doi.org/10.4067/S0718-58392022000200199

Mafongoya PL, Barak P, Reed JD (2000) Carbon, nitrogen and phosphorus mineralization of tree leaves and manure. Biol Fertil Soils. https://doi.org/10.1007/s003740050007

Marouani E, Ziadi N, Lévesque V, Benzina K, Bouslimi N, B. and, Koubaa A (2021) Mitigation of CO_2, CH_4 and N_2O from Acidic Clayey Soil Amended with Fertilizer Pellets Based on Alkaline Organic Residues. Waste Biomass Valorization. https://doi.org/10.1007/s12649-020-01276-y

Mbava N, Zengeni R, Muchaonyerwa P (2025) Carbon dioxide emissions and nitrogen and phosphorus mineralization patterns from soil amended with shoot and root residues of different wheat genotypes. Front Agron. https://doi.org/10.3389/fagro.2025.1576878

McMillen MS, Mahama AA, Sibiya J, Lübberstedt T, Suza WP (2022) Improving drought tolerance in maize: Tools and techniques. Front Genet. https://doi.org/10.3389/fgene.2022.1001001

Meng F, Dungait JA, Xu X, Bol R, Zhang X, Wu W (2017) Coupled incorporation of maize (*Zea mays* L.) straw with nitrogen fertilizer increased soil organic carbon in fluvic cambisol. Geoderma. https://doi.org/10.1016/j.geoderma.2016.09.010

Nelissen V, Rütting T, Huygens D, Staelens J, Ruysschaert G, Boeckx P (2012) Maize biochars accelerate short-term soil nitrogen dynamics in a loamy sand soil. Soil Biol Biochem. https://doi.org/10.1016/j.soilbio.2012.05.019

Nguyen BT, Phan BT, Nguyen TX, Nguyen VN, Van Tran T, Bach QV (2020) Contrastive nutrient leaching from two differently textured paddy soils as influenced by biochar addition. J Soils Sediments. https://doi.org/10.1007/s11368-019-02366-8

Non-Affiliated Soil Analysis Work Committee (1990) Handbook of standard soil testing methods for advisory purposes. Soil Science Society of South Africa, Pretoria

Nord A, Snapp S, Traore B (2022) Current knowledge on practices targeting soil fertility and agricultural land rehabilitation in the Sahel. A review. Agron Sustain Dev. https://doi.org/10.1007/s13593-022-00808-1

Novak JM, Busscher WJ, Watts DW, Laird DA, Ahmedna MA, Niandou MA (2010) Short-term CO2 mineralization after additions of biochar and switchgrass to a typic kandiudult. Geoderma. https://doi.org/10.1016/j.geoderma.2009.10.014

Ntonta S, Zengeni R, Muchaonyerwa P, Chaplot V (2024) Variability in decomposition rate of sorghum cultivar residues linked to lignin content. Rhizosphere. https://doi.org/10.1016/j.rhisph.2024.100850

ODUGBENRO GO, Yankun SUN, Zhihua LIU (2018) Effects of biochar on carbon pool, N mineralization, microbial biomass and microbial respiration from mollisol. Afr J Agric Res DOI. https://doi.org/10.5897/AJAR2018.13505

Okalebo JR, Gathua KW, Woomer PL (2002) Laboratory methods of soil and plant analysis: a working manual second edition. Sacred Africa, Nairobi.

Partey ST, Zougmore RB, Thevathasan NV, Preziosi RF (2019) Effects of plant residue decomposition on soil N availability, microbial biomass and β-glucosidase activity during soil fertility improvement in Ghana. Pedosphere. https://doi.org/10.1016/S10020160(17)60433-8

Payne GT, Moore CB, Griffis SE, Autry CW (2011) Multilevel challenges and opportunities in social capital research. J Managt. https://doi.org/10.1177/0149206310372413

Peng XYLL, Ye LL, Wang CH, Zhou H, Sun B (2011) Temperature- and duration-dependent rice straw-derived biochar: Characteristics and its effects on soil properties of an Ultisol in southern China. Soil Till Res. https://doi.org/10.1016/j.still.2011.01.002

Powlson DS, Bhogal A, Chambers BJ, Coleman K, Macdonald AJ, Goulding KWT, Whitmore AP (2012) The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: a case study. Agric Ecosyst Environ. https://doi.org/10.1016/j.agee.2011.10.004

Prasad VR, Govindaraj M, Djanaguiraman M, Djalovic I, Shailani A, Rawat N, Singla-Pareek SL, Pareek A, Prasad PV (2021) Drought and high temperature stress in sorghum: Physiological, genetic, and molecular insights and breeding approaches. Int J Mol Sci. https://doi.org/10.3390/ijms22189826

Purakayastha TJ, Bera T, Bhaduri D, Sarkar B, Mandal S, Wade P, Kumari S, Biswas S, Menon M, Pathak H, Tsang DC (2019) A review on biochar modulated soil condition improvements and nutrient dynamics concerning crop yields: pathways to climate change mitigation and global food security. Chemosphere. https://doi.org/10.1016/j.chemosphere.2019.03.170

Rayment GE, Lyons DJ (2011) Soil chemical methods: Australasia (vol 3). CSIRO publishing. https://doi.org/10.1071/9780643101364

Redin M, Recous S, Aita C, Dietrich G, Skolaude AC, Ludke WH, Schmatz R, Giacomini SJ (2014) How the chemical composition and heterogeneity of crop residue mixtures decomposing at the soil surface affects C and N mineralization. Soil Biol Biochem. https://doi.org/10.1016/j.soilbio.2014.07.014

Rizwan M, Murtaza G, Zulfiqar F, Moosa A, Iqbal R, Ahmed Z, Irshad S, Khan I, Li T, Chen J, Zhang M (2023) Sustainable manufacture and application of biochar to improve soil properties and remediate soil contaminated with organic impurities: a systematic review. Front Environ Sci. https://doi.org/10.3389/fenvs.2023.1277240

Schmatz R, Recous S, Aita C, Tahir MM, Schu AL, Chaves B, Giacomini SJ (2017) Crop residue quality and soil type influence the priming effect but not the fate of crop residue C. Plant Soil. https://doi.org/10.1007/s11104-016-3120-x

Shetty R, Prakash NB (2020) Effect of different biochars on acid soil and growth parameters of rice plants under aluminium toxicity. Sci Rep. https://doi.org/10.1038/s41598-020-69262-x

Silva ICBD, Basílio JJN, Fernandes LA, Colen F, Sampaio RA, Frazão LA (2017) Biochar from different residues on soil properties and common bean production. Scientia Agricola. https://doi.org/10.1590/1678-992x-2016-0242

Soil Classification Working Group. (1991). Soil classification: a taxonomic system for South Africa (vol 15). Department of Agricultural Development.

Thomazini A, Spokas K, Hall K, Ippolito J, Lentz R, Novak J (2015) Ghg impacts of biochar: predictability for the same biochar. Agric Ecosyst Environ. https://doi.org/10.1016/j.agee.2015.04.012

van Midden C, Harris J, Shaw L, Sizmur T, Morgan H, Pawlett M (2024) Immobilisation of anaerobic digestate supplied nitrogen into soil microbial biomass is dependent on lability of high organic carbon materials additives. Front Sustain Food Syst. https://doi.org/10.3389/fsufs.2024.1356469

Van Soest PV, Robertson JB, Lewis BA (1991) Methods for dietary fiber, neutral detergent fiber, and nonstarch polysaccharides in relation to animal nutrition. J Dairy Sci. https://doi.org/10.3168/jds.S0022-0302(91)78551-2

Vilakazi SP, Muchaonyerwa P, Buthelezi-Dube NN (2023) Characteristics and liming potential of biochar types from potato waste and pine-bark. PLoS One. https://doi.org/10.1371/journal.pone.0282011

Wang H, Ren T, Yang H, Feng Y, Feng H, Liu G, Shi H (2020) Research and application of biochar in soil CO_2 emission, fertility, and microorganisms: A sustainable solution to solve China's agricultural straw burning problem. Sustainability. https://doi.org/10.3390/su12051922

Wang Q, Yuan J, Yang X, Han X, Lan Y, Cao D, Chen W (2022) Responses of soil respiration and C sequestration efficiency to biochar amendment in maize field of Northeast China. Soil Tillage Res. https://doi.org/10.1016/j.still.2022.105442

Watanabe S, Sato S (2015) Priming effect of bamboo (*Phyllostanchys edulis Carriere)* biochar application in a soil amended with

legume. Soil Sci Plant Nutr DOI. https://doi.org/10.1080/00380768.2015.1105112
Whitman T, Enders A, Lehmann J (2012) Pyrogenic carbon additions to soil counteract positive priming of soil carbon mineralization by plants. Soil Biol Biochem. https://doi.org/10.1016/j.soilbio.2014.02.009
Working Group IUSS (2014) World reference base for soil resources 2014. International soil classification system for naming soils and creating legends for soil maps. https://doi.org/10.1007/978-3-319-24409-9_25
Xiaoping Tian, Lei Wang, Yahong Hou, Han WANG, Jihua Wu (2019) Responses of soil microbial community structure and activity to incorporation of straws and straw biochars and their effects on soil respiration and soil organic carbon turnover. Pedosphere. https://doi.org/10.1016/S1002-0160(19)60813-1
Yao Y, Gao B, Zhang M, Inyang M, Zimmerman AR (2012) Effect of biochar amendment on sorption and leaching of nitrate, ammonium, and phosphate in a sandy soil. Chemosphere. https://doi.org/10.1016/j.chemosphere.2012.06.002
You M, Li LJ, Tian Q, He P, He G, Hao XX, Horwath WR (2020) Residue decomposition and priming of soil organic carbon following different NPK fertilizer histories. Soil Sci Soc Am J. https://doi.org/10.1002/saj2.20142
Zavalloni C, Alberti G, Biasiol S, Delle Vedove G, Fornasier F, Liu J, Peressotti A (2011) Microbial mineralization of biochar and wheat straw mixture in soil: a short-term study. Appl Soil Ecol. https://doi.org/10.1016/j.apsoil.2011.07.012
Zhai L, CaiJi Z, Liu J, Wang H, Ren T, Gai X, Liu H (2015) Short-term effects of maize residue biochar on phosphorus availability in two soils with different phosphorus sorption capacities. Biol Fertil Soils. https://doi.org/10.1007/s00374-014-0954-3
Zhang P, Chen X, Wei T, Yang Z, Jia Z, Yang B, Han Q, Ren X (2016) Effects of straw incorporation on the soil nutrient contents, enzyme activities, and crop yield in a semiarid region of China. Soil Tillage Res. https://doi.org/10.1016/j.still.2016.02.006
Zhao Y, Lin S, Liu Y, Li G, Wang J, Butterbach-Bahl K (2020) Application of mixed straw and biochar meets plant demand of carbon dioxide and increases soil carbon storage in sunken solar greenhouse vegetable production. SOIL USE MANAGE. https://doi.org/10.1111/sum.12579
Zheng H, Wang Z, Deng X, Herbert S, Xing B (2013) Impacts of adding biochar on nitrogen retention and bioavailability in agricultural soil. Geoderma. https://doi.org/10.1016/j.geoderma.2013.04.018