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# Biochar amendments mitigate soil greenhouse gas emissions by shifted soil properties, enzyme activities, and nitrogen cycling processes

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## Abstract

Amending agricultural soils with biochar has emerged as a promising approach to sustainably mitigate soil greenhouse gas (GHG) emissions, but information on mechanisms underlying biochar-induced emission dynamics of GHGs remains limited. Our study aimed to comprehensively explore the significance of biochar mediated changes of soil properties, enzyme activities, and nitrogen cycling processes, for mitigating GHG emissions and lowering the concomitant global warming potential (GWP). For this purpose, we conducted a meta-analysis of 78 published studies investigating the effects of biochar amendments on soil properties in agricultural systems worldwide. Biochar amendment significantly increased soil porosity, moisture and total carbon (+ 57% to + 62%), but also soil organic carbon (+ 24%) and total nitrogen (+ 26%). Additionally, salinity, cation exchange capacity, base saturation, and C:N ratio increased (+ 20% to + 29%), while bulk density, labile organic carbon, ammonium- and ammonia-nitrogen, nitrate- and nitrite-nitrogen decreased (− 8% to − 38%). Biochar negatively affected the activities of key soil enzymes, such as  $\beta$ -glucosidase, N-acetyl glucose-aminidase, and acid phosphatase (− 14% to − 34%), enhanced ammonification and biological nitrogen fixation (+ 11% to + 13%), while suppressing denitrification and nitrification (− 6% to − 12%). These shifts contributed to significant reductions in GHG emissions: carbon dioxide (− 24% on average), methane (− 36% to − 22%), and nitrous oxide (− 33% to − 39%). Consequently, global warming potential (GWP) was lowered, with the extent of reduction depending on biochar properties and management practices. High pyrolysis temperatures (> 400 °C), long-term application (100-year scale), and higher biochar doses ( $\geq 40 \text{ t}\cdot\text{ha}^{-1}$ ) induced the greatest reductions in GWP (− 83% to − 66%). Among cropping systems, rice fields benefited most from biochar application (lowest greenhouse gas emission intensity, − 53%), while maize systems contributed most to GWP among analyzed crops. Therefore, strategic biochar deployment, optimized for specific cropping systems and pyrolysis conditions, offers a powerful and scalable pathway towards climate-positive agriculture.

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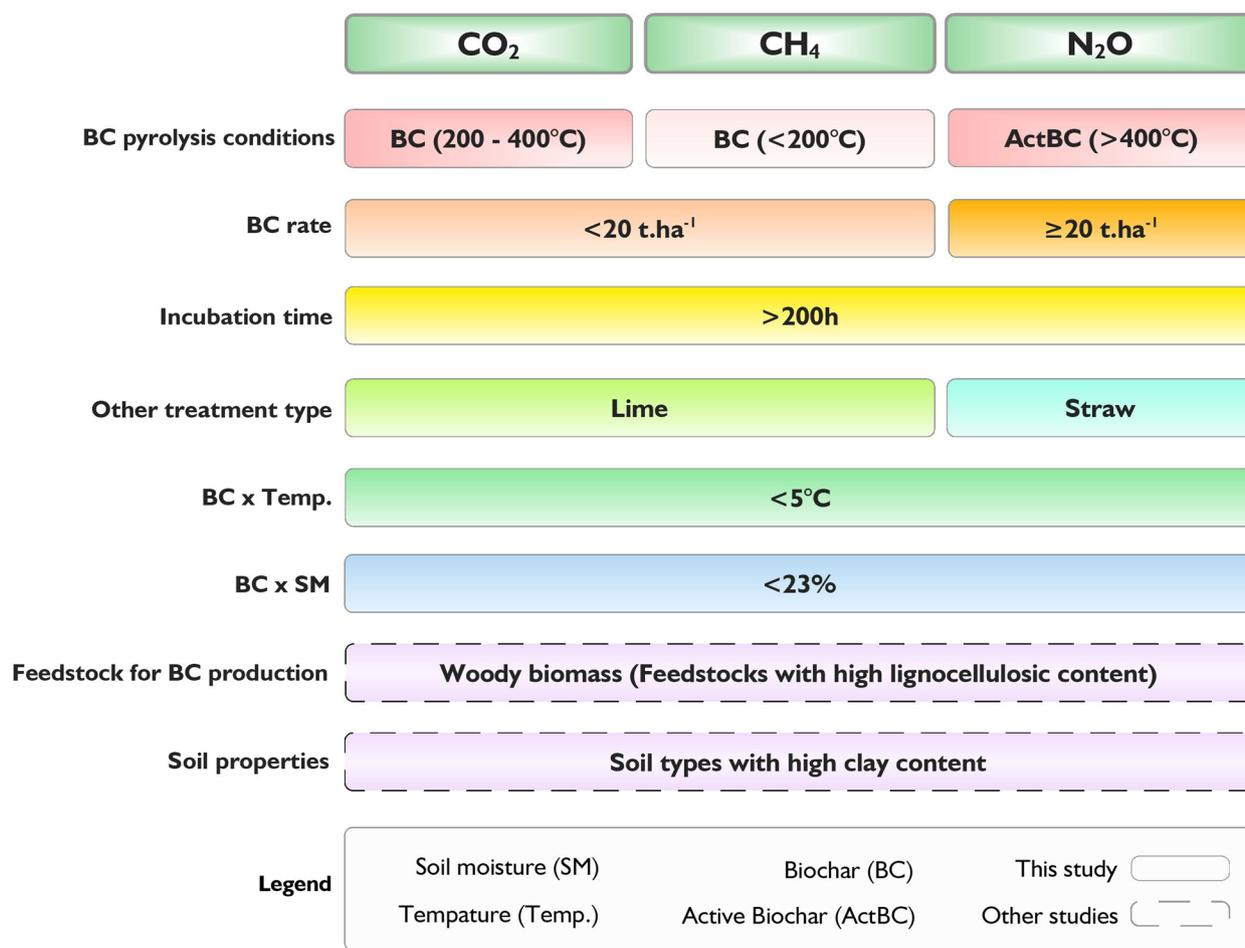
**Highlights**

- Biochar lowers GHG emissions by improving soil properties and inhibiting enzyme activities.
- Biochar favors ammonification and N-fixation while suppressing denitrification and nitrification.
- GHG mitigation occurs with high-temperature (>400°C) treated biochar and at doses ≥40 t·ha<sup>-1</sup>.
- Biochar can mitigate GHG emissions by 24%-39% in various agricultural systems.

**Keywords** Biochar amendments, Greenhouse gas emissions, Soil properties, Enzyme activities, Nitrogen cycling processes

**Graphical Abstract**

## GHG mitigation efficiency conditions



**1 Introduction**

Biochar has recently garnered attention as a soil amendment due to its potential to improve agricultural productivity and address environmental challenges. Recent

studies have demonstrated its beneficial impacts on soil fertility, crop biomass production, litter decomposition rates, and carbon (C) sequestration (Ding et al. 2016; Jeffery et al. 2011; Lorenz and Lal 2014). As a valuable soil

amendment, biochar enhances soil properties by retaining water, nutrients, and beneficial microorganisms, thereby improving soil moisture and nutrient availability (Ball 2013; Cai et al. 2018; Sohi et al. 2010). Additionally, it enhances soil structure, porosity, cation exchange capacity, and pH, increasing overall fertility (Guo 2020; Sun et al. 2022).

Biochar also plays a critical role in nitrogen cycling by providing habitats for microorganisms that convert organic nitrogen compounds to ammonium ions (Hagemann et al. 2016). It enhances denitrification by promoting bacterial growth and facilitating the conversion of nitrate to dinitrogen ( $N_2$ ) instead of nitrous oxide (Clough and Condon 2010). Furthermore, biochar adsorbs and stabilizes nitrogen, reducing the risk of leaching and the need for synthetic fertilizer application (Zheng et al. 2013). These changes in nitrogen cycling have significant implications for soil greenhouse gas (GHG) emissions, with biochar acting as a stable carbon sink that prevents nitrogen from converting to  $N_2O$  (Horák et al. 2021).

Despite these benefits, our understanding of the mechanisms behind biochar-induced GHG emissions, such as carbon dioxide ( $CO_2$ ), methane ( $CH_4$ ), and nitrous oxide ( $N_2O$ ), is limited. Previous studies have reported conflicting results: some indicate that biochar reduces GHG emissions (Eykelbosh et al. 2014), while others show stimulatory effects (Zhang et al. 2010) or no significant impacts (Scheer et al. 2011). These discrepancies arise from various factors, including differences in biochar processing, application rates, incubation duration, and physicochemical properties (Ahmad et al. 2021; Sohi et al. 2010; Subedi et al. 2016).

In addition to its role in nitrogen cycling, amending soils with biochar can also reduce  $CO_2$  emissions by enhancing carbon sequestration (Mukherjee et al. 2014). Its high carbon content and resistance to decomposition make it an effective long-term carbon sink. Biochar may further reduce  $CH_4$  emissions by altering soil properties and enhancing microbial activity (Cai et al. 2018). However, in acidic soils, biochar helps neutralize pH, thereby inhibiting the growth of methanotrophic bacteria that consume methane (Guo et al. 2016). Moreover, biochar can significantly reduce  $N_2O$  emissions by increasing soil pH, improving water-holding capacity, and enhancing nitrogen retention (Verhoeven et al. 2017).

Soil enzyme activities also significantly influence GHG emissions by breaking down organic matter (Liu et al. 2019; Pandey et al. 2020). Adding biochar can reduce  $\beta$ -glucosidase activity (Foster et al. 2018), potentially leading to increased organic matter decomposition and  $CO_2$  release (Mariscal-Sancho et al. 2010). Liu et al. (2022) found positive relationships between  $CH_4$  emissions and both, acid phosphatase (ACP) and

$\beta$ -cellobiohydrolase (CBH) activities. Additionally, biochar amendment enhanced soil leucine aminopeptidase (LAP) activity, indicating increased microbial activity in protein degradation and potential reductions in  $CH_4$  emissions (Yeboah et al. 2018; Zhang et al. 2024). Furthermore, Mariscal-Sancho et al. (2010) reported a positive association between  $\beta$ -glucosidase activity and  $N_2O$  emissions, as the breakdown of carbohydrates releases carbon and nitrogen compounds that contribute to  $N_2O$  emissions (Sial et al. 2022).

Despite promising findings regarding biochar application, a systematic synthesis of outcomes related to its use in climate-smart agriculture is lacking, hindering the development of universal guidelines. Critical questions remain unresolved, including how biochar properties (e.g., feedstock and pyrolysis temperature) interact with soil conditions to influence GHG fluxes. Furthermore, optimal thresholds for biochar application rates and incubation times to maximize GHG mitigation need to be determined, as well as which cropping systems (e.g., rice versus maize) benefit most from biochar-induced GHG reductions.

To address these gaps, this study aims to answer the following research questions: How do biochar properties (e.g., feedstock, pyrolysis temperature) and soil conditions interact to influence GHG emissions? What are the optimal biochar application rates and incubation times for maximizing GHG mitigation? Which cropping systems benefit most from biochar-induced GHG reductions? By investigating these questions, this study seeks to provide a comprehensive understanding of the impacts of biochar amendments on soil properties, enzyme activities, and nitrogen cycling processes, and how these interactions collectively drive GHG emissions. Understanding these relationships can provide valuable insights into the roles of soil biochar amendments in sustainable agriculture and climate change mitigation. Ultimately, this study aims to transform fragmented knowledge into a predictive framework for targeted biochar deployment to meet climate goals, unifying mechanistic insights with practical scalability in sustainable agriculture.

## 2 Materials and methods

### 2.1 Data collection and classification of GHG emissions

We searched for information on GHG emissions in peer-reviewed journal publications, focusing on  $CO_2$ ,  $CH_4$ , and  $N_2O$  emissions under biochar amendment. The search was conducted using various combinations of keywords such as GHG emissions, biochar amendment, lime application, N application forms (including urea and  $NH_4NO_3$ ), and straw via the ISI Web of Science and Google Scholar, covering publications up until October 2023. The search strategy included the following

keywords or terms: (biochar OR "bio-char") AND ("greenhouse gas\*" OR GHG OR CO<sub>2</sub> OR "carbon dioxide" OR CH<sub>4</sub> OR "methane" OR N<sub>2</sub>O OR "nitrous oxide") AND ("soil property\*" OR "soil characteristic\*" OR "soil parameter\*") AND ("enzyme activity\*" OR "soil enzyme\*" OR "β-glucosidase" OR "phosphatase" OR "aminopeptidase") AND ("nitrogen cycling" OR "N cycling" OR "nitrification" OR "denitrification" OR "ammonification" OR "nitrogen fixation") AND ("agriculture\*" OR "cropping system\*" OR "farmland" OR "agroecosystem\*").

A total of 78 publications were selected based on specific criteria: experiments conducted in agricultural ecosystems, consistent initial environmental conditions and soil properties for both groups, availability of GHG emissions data, experiments with at least one pair of control and treatment data, and the ability to extract means, standard deviations/errors, and sample sizes directly from text, tables, or digitized graphs (Fig. S1). The flow diagram for study inclusion and exclusion reported a total of 867 included studies and 789 excluded studies, along with the reasons for exclusion (Fig. S1). To collect data on GHG emissions, we carefully extracted information from the text, tables, figures, and appendices of the selected publications (Fig. S1). In cases where results were presented graphically, we employed the GetData Graph Digitizer v.2.20 software to extract numerical values. The selected publications also provided information on pH, organic C, inorganic C, soil temperature (Ts), soil moisture (SM), bulk density (BD), water-holding capacity (WHC), dissolved organic carbon (DOC), labile organic carbon (LOC), microbial biomass carbon (MBC), dissolved organic nitrogen (DON), nitrogen dioxide (NO<sub>2</sub>), nitrate (NO<sub>3</sub>), ammonia (NH<sub>3</sub>), and ammonium (NH<sub>4</sub>) emissions to assess the main factors controlling GHG variables. When this information was not reported, we sourced it from the database at <http://www.worldclim.org/> based on location data, such as latitude and longitude.

To enhance clarity and understanding, a flowchart or diagram was included to visually represent the data collection and analysis process. We followed PRISMA guidelines to ensure a comprehensive and systematic approach in our review process (Dickson and Yeung 2022; Selçuk 2019). By adhering to these guidelines, we enhanced the transparency and reproducibility of our research, which facilitated a thorough assessment of the available literature (Table S1). This not only strengthened the validity of our findings but also provided a clear framework for other researchers seeking to replicate or build upon our work. Data classification is a crucial step in the meta-analysis process, as

it involves organizing and categorizing the various data sets that will be analyzed. Assessing both biochar and activated biochar (a form of biochar that has undergone additional processing to increase its surface area, porosity, and adsorption capacity) for soil GHG emissions is important to understand their differential impacts on microbial activity, carbon sequestration, methane oxidation, nitrous oxide emissions, and long-term stability.

To facilitate this analysis, we categorized the variables as follows: Biochar (BC) processing: BC (<200 °C), BC (200–400 °C), BC (>400 °C), activated biochar (ActBC) (<200 °C), ActBC (200–400 °C), ActBC (>400 °C). Biochar rate: <20 t·ha<sup>-1</sup> (Low biochar; LBC) and ≥20 t·ha<sup>-1</sup> (High biochar; HBC). Treatment type: lime, NH<sub>4</sub>NO<sub>3</sub>, biochar x nitrogen (BIN), urea, and straw. Feedstock: wood residue and crop residue. Incubation time: <100 h, 100–200 h, >200 h. Biochar interaction with temperature: BC x <5 °C, BC x 5–10 °C, BC x >10 °C, and biochar interaction with SM: BC x <23%, and BC x ≥23%. Active biochar refers to biochar that has been chemically or physically modified after pyrolysis to enhance its functional properties, such as surface area, porosity, cation exchange capacity, nutrient retention, and catalytic activity.

## 2.2 Classification of soil property data

The variables related to soil properties were identified and classified into three categories: soil physical properties, soil chemical properties, and soil biological properties. For *soil physical properties*, we used the variables SM, BD, soil porosity, soil temperature (Ts), WHC, and aggregate stability (Agg. sta.), particularly water-stable aggregates (WSA) and mean weight diameter (MWD). *Soil chemical properties* included soil pH, DOC, soil organic carbon (SOC), total nitrogen (Total N), DON, nitrate-nitrogen (NO<sub>3</sub><sup>-</sup>-N), ammonium-nitrogen (NH<sub>4</sub><sup>+</sup>-N), nitrite-nitrogen (NO<sub>2</sub>-N), total carbon (Total C), carbon-to-nitrogen ratio (C:N ratio), carbon-to-phosphorus ratio (C:P ratio), nitrogen-to-phosphorus ratio (N:P ratio), ammonia-nitrogen (NH<sub>3</sub>-N), salinity, base saturation, LOC, the solubility of N<sub>2</sub> and cation exchange capacity (CEC). Lastly, two variables were related to *soil biological properties*, i.e., MBC and phospholipid fatty acid (PLFA).

## 2.3 Classification of soil enzyme activities and N cycling processes

We selected enzymes involved in organic matter decomposition, nutrient availability, substrate utilization, and microbial community dynamics, all of which contribute to GHG emissions from soils. A total of five soil enzyme

activities were selected, including  $\beta$ -glucosidase ( $\beta$ G), N-acetyl glucosamine (NAG), ACP,  $\beta$ -cellobiohydrolase (CBH), and leucine aminopeptidase (LAP). Nitrogen cycling in the soil is closely linked to GHG emissions. Therefore, five nitrogen cycling processes involved in the availability and transformations of N compounds in soil were included, *i.e.*, ammonification, denitrification, N-fixation, N mineralization, and nitrification.

### 2.4 Data analysis

To assess potential publication bias, we constructed funnel plots for effect sizes and conducted Egger’s regression tests. The funnel plot and statistical analysis were based on a model estimated using the restricted maximum likelihood (ML) method, focusing on fixed and random effects (Table S2 and Fig. S2). If significant asymmetry was detected, we applied the trim-and-fill method to estimate corrected effect sizes. The omnibus test of model coefficients yields a highly significant ( $p < 0.001$ ), indicating that the model’s coefficients are statistically robust and the model fits the data well (Table S2a). Additionally, the test of residual heterogeneity also shows a significant ( $p < 0.001$ ), suggesting substantial unexplained variability in the data. The intercept of the model is estimated at 0.453 with a standard error of 0.018, and it is highly significant ( $p < 0.001$ ), further supporting the model’s validity (Table S2b). However, the residual heterogeneity estimates reveal considerable between-study variance ( $\tau^2 = 0.074$ ) and a very high  $I^2$  value (99.75%) (Table S2c), indicating that a large proportion of the observed variance is due to heterogeneity rather than chance. This suggests that the model is statistically significant and fits well. Egger’s regression test for publication bias yielded a p-value greater than 0.05, indicating no significant publication bias in the meta-analysis.

#### 2.4.1 Meta-analysis

To assess the impacts of biochar on GHG emissions, we used the method by Hedges et al. (1999), applying the natural log-transformed response ratio (RR) in a meta-analysis. The RR was weighted by sample size and variance to ensure result validity.

$$RR = \ln \left( \frac{\bar{X}_t}{\bar{X}_c} \right) = \ln(\bar{X}_t) - \ln(\bar{X}_c) \tag{1}$$

$X_t$  represents the observed values of a chosen variable in the mixture, while  $X_c$  denotes the expected value of the mixture. The meta-analyses were conducted using the METAWIN software version 2.1 (Sinauer Associates, Inc. Sunderland, MA, USA). The estimation of the RR was carried out as follows:

$$v = \frac{S_t^2}{n_t X_t^2} + \frac{S_c^2}{n_c X_c^2} \tag{2}$$

where  $n_t$  and  $n_c$  represent the sample sizes of the experimental and control groups, respectively.  $S_t$  and  $S_c$  denote the standard deviations, while  $X_t$  and  $X_c$  indicate the mean response values in the experimental and control groups, respectively. In cases where literature sources provided the standard error instead of the standard deviation (SD), we recalculated SD using the following approach:

$$SD = SE \times \sqrt{N} \tag{3}$$

where  $N$  is the number of replications.

We calculated the weighting factor ( $W_{ij}$ ) and standard error (SE) of  $\ln(RR)$  using the equations provided below:

$$W_{ij} = \sqrt{\frac{1}{V_{ij}}} \tag{4}$$

$$S(\ln(RR)) = \sqrt{\frac{1}{\sum_{i=1}^m \sum_{j=1}^{ki} W_{ij}}} \tag{5}$$

We calculated the 95% confidence interval (95% CI) using the following equation:

$$95\%CI = \ln(RR) \pm 1.96S(\ln(RR)), \tag{6}$$

$$95\%CI = \ln RR_{++} \pm 1.96 \times \sqrt{\frac{1}{\sum_{i=1}^j w_i}} \tag{7}$$

The percentage change of the variables caused by biochar amendment was calculated as follows:

$$\text{Chang}(\%) = [\exp(RR_{++}) - 1] \times 100\% \tag{8}$$

#### 2.4.2 Statistical modeling approach

We conducted a comprehensive mixed-effects meta-regression analysis using JMP Pro version 16 to evaluate factors influencing biochar’s impact on GHG ( $\text{CO}_2$ ,  $\text{CH}_4$ , and  $\text{N}_2\text{O}$ ) emissions. Our analysis incorporated multiple covariates including soil properties (SM, MBC, Ts, aggregate stability, salinity, BD, porosity, pH, PLFA, CEC), biochar characteristics (processing method, application rate, duration), and environmental factors (incubation time, temperature).

#### 2.4.3 Covariate selection and model development

To ensure robust model performance, we implemented a rigorous two-stage covariate selection procedure. First,

we assessed multicollinearity using Variance Inflation Factors (VIF), excluding variables with  $VIF \geq 5$  (Petrini et al. 2012) to prevent distortion of parameter estimates. This conservative threshold aligns with established practices and helps maintain the validity of statistical inferences (Radočaj et al. 2023). Second, we applied recursive feature elimination (RFE), iteratively removing the least significant covariates based on model coefficients to optimize predictive performance while minimizing multicollinearity (Huang et al. 2024). This approach enhances model interpretability and robustness, particularly for high-dimensional datasets (Ngaba et al. 2024; Rajalahti et al. 2021).

#### 2.4.4 Model specification and validation

The final models employed restricted maximum likelihood (REML) estimation, incorporating both fixed and random effects to account for variability in the data. The fixed effects included selected covariates that met the significance threshold, defined as having an absolute beta coefficient ( $|\beta|$ ) of 0.8 or greater. This ensured that only the most relevant predictors were included in the model. For the random effects, study-level random intercepts were used to account for between-study heterogeneity, allowing the model to capture variations specific to individual studies. Model validation was conducted through several rigorous steps. First, the normality of the residuals was verified using Q-Q plots, which are graphical tools that help assess whether the residuals follow a normal distribution a key assumption in many statistical models. Additionally, sensitivity analyses were performed by excluding potential outlier studies to evaluate their impact on the model's results, ensuring robustness and reliability. Finally, the models were evaluated based on the explained variance, with  $R^2$  values indicating the proportion of variability in the data that was accounted for by the models. The  $R^2$  values were 0.921 for  $CO_2$  and 0.534 for  $N_2O$ , suggesting that the models explained a substantial portion of the variance in these GHGs, with particularly strong explanatory power for  $CO_2$ .

This dual approach of VIF screening and RFE optimization (Baumeister et al. 2024) achieved our three primary objectives: (i) improved computational efficiency, (ii) mitigated overfitting risk, and (iii) ensured unbiased variable significance evaluation, while maintaining ecological relevance through our standardized coefficient threshold. The methodology provides a robust framework for analyzing complex biochar-GHG interactions across diverse agricultural systems.

#### 2.4.5 Global warming potential and greenhouse gas emission intensity calculation

The following equations were applied to calculate GWP as means to quantify GHG effects caused by biochar amendment at 20 and 100 years, if data of GWP (control and treatment) were not available.

$$GWP_{20}(tCO_2 - eq ha^{-1}) = \left[ \frac{(84 \times CH_4 + 264 \times N_2O)}{1000} \right]_x, \quad (9)$$

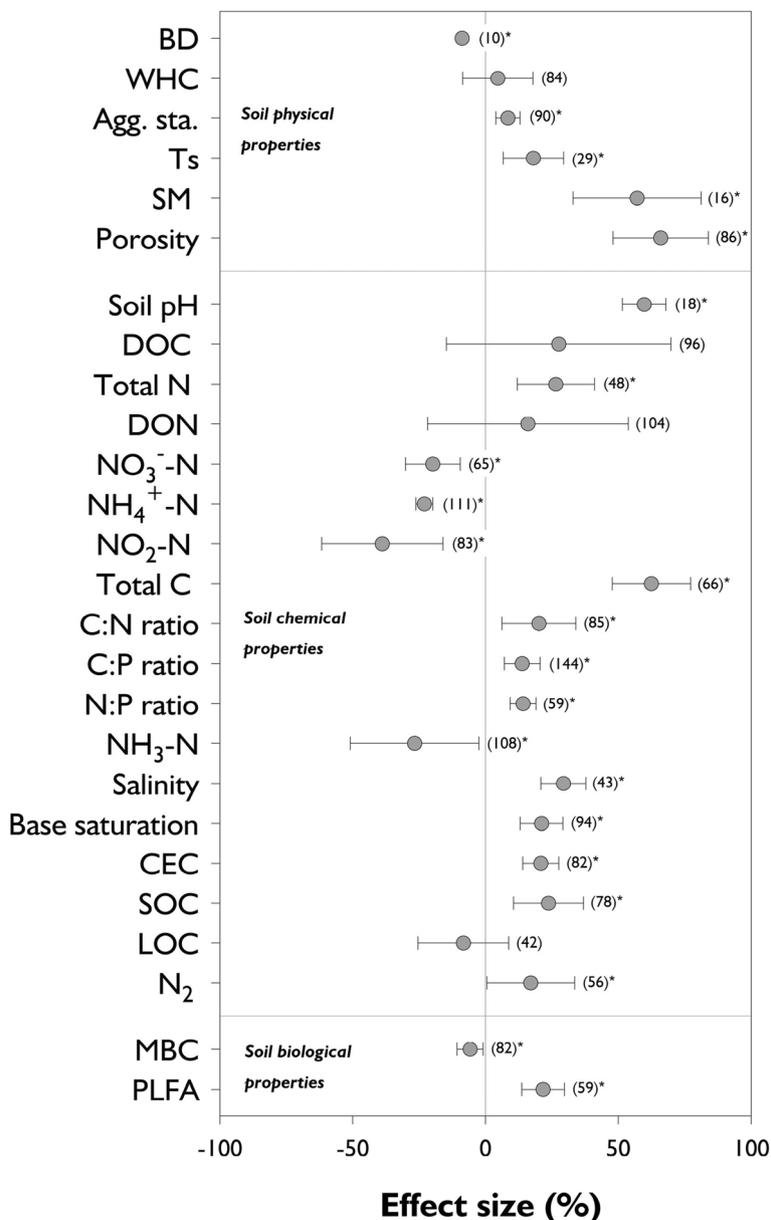
$$GWP_{100}(tCO_2 - eq ha^{-1}) = \left[ \frac{(28 \times CH_4 + 265 \times N_2O)}{1000} \right]_x, \quad (10)$$

where  $x$  represents the control or treatment (biochar application only); The changes in emissions of  $CH_4$  and  $N_2O$  ( $kg ha^{-1}$ ) were converted into " $CO_2$ -equivalents"; 84 and 28 indicate the GWP of  $CH_4$  while 264 and 265 indicate the GWP of  $N_2O$  per unit mass at 20 and 100 years scale (IPCC 2013). The division by 1000 in Eq. (9) serves to convert the final GWP value from kilograms (kg) to metric tons (t) of  $CO_2$ -equivalents per hectare (ha). According to Masson-Delmotte et al. (2021), GWP calculations follow the IPCC's "*Kyoto basket*" approach for agricultural soils, prioritizing  $CH_4$  and  $N_2O$  because of their disproportionate warming impacts. In contrast, the  $CO_2$  emissions from biochar, which are context-dependent (e.g., feedstock pyrolysis temperature), are excluded from GWP calculations due to their biogenic origin.

To identify opportunities for emission reductions and promote resource efficiency, we calculated the greenhouse gas emission intensity (GHGI) using the following equation.

$$GHGI = GWP/yield, \quad (11)$$

where GHGI is the GHGI of crop yield at 100 years induced by biochar ( $kg CO_2$ -eq  $ha^{-1}$ ). We selected the yields of four crops, i.e., rice, sunflowers, wheat, and maize, because of their economic importance, high global production, and consumption. These crops play a crucial role in food security and agricultural sustainability, serving as staple foods for billions of people around the world (González et al. 2021; Ray et al. 2012). Rice, for instance, is a primary source of energy for billions of people, particularly in Asia (Landi et al. 2017), while sunflowers contribute significantly to the oil market (Yu and Shang 2018). Wheat is a cornerstone grain for many countries, forming the basis of numerous food products, and maize is integral for human consumption as well as for livestock feed (Li et al. 2023). Additionally, they contribute significantly to the livelihoods of farmers and rural communities, providing income and employment opportunities (Makate et al. 2016). The selection of these crops also reflects their versatility in various climatic conditions



**Fig. 1** Effects of biochar amendment on soil properties, nitrogen and carbon contents and emissions. Black bars represent 95% confidence intervals. The different letters in parentheses indicate the number of observations. A dashed vertical line is drawn at a mean effect size of 0. \*Significant at the 0.05 level. Non-biochar (BC), active-biochar (ActBC), soil temperature (Ts), phospholipid fatty acid (PLFA), soil moisture (SM), bulk density (BD), water-holding capacity (WHC), dissolved organic carbon (DOC), labile organic carbon (LOC), microbial biomass carbon (MBC), and dissolved organic nitrogen (DON), nitrogen dioxide (NO<sub>2</sub>), nitrate (NO<sub>3</sub>), ammonia (NH<sub>3</sub>) and ammonium (NH<sub>4</sub>) emissions

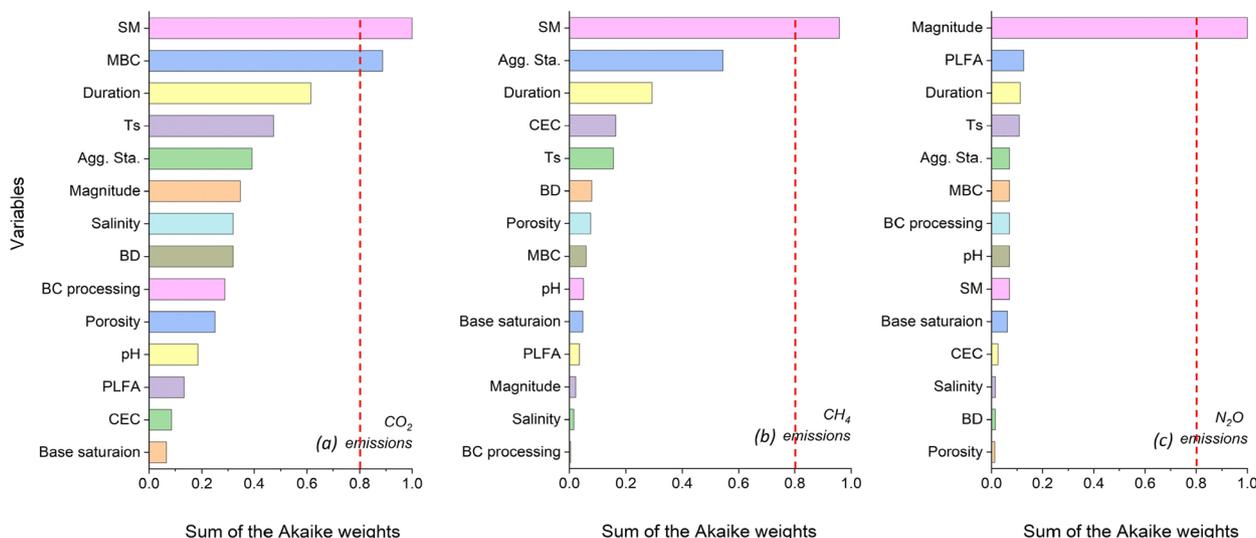
and their potential for contributing to the diversification of agricultural systems (Ghosh 2021; Neogi and Ghosh 2022).

### 3 Results

#### 3.1 Biochar amendment affects soil properties

Biochar amendment significantly altered soil properties (Fig. 1). It increased soil porosity (+65%, CI: 58%–72%),

SM (+57%, CI: 49%–65%), temperature (+18%, CI: 12%–24%), and aggregate stability (+4%, CI: 1%–7%) ( $p < 0.01$ ). Conversely, it decreased BD (–8%, CI: –11% to –5%) (Fig. 1;  $p < 0.01$ ) and had no significant effect on WHC. Chemical properties improved with increases in total C (+62%, CI: 55%–69%), SOC (+24%, CI: 18%–30%), DOC (+27%, CI: 20%–34%), total N (+26%, CI: 19%–33%), DON (+16%, CI: 9%–23%), N<sub>2</sub> (+17%, CI: 10%–24%), pH



**Fig. 2** Relative importance of variables regulating the effects of biochar amendment on (a) carbon dioxide (CO<sub>2</sub>), (b) methane (CH<sub>4</sub>), and (c) nitrous oxide (N<sub>2</sub>O) emissions. The results are based on studies that simultaneously reported the CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions responses. SM-induced shifts in CO<sub>2</sub> emissions under biochar addition are negatively correlated with CO<sub>2</sub> emissions ( $R^2=0.921$ ;  $p<0.001$ ;  $n=39$ ). Biochar magnitude-induced a positive shift in N<sub>2</sub>O emissions ( $R^2=0.534$ ;  $p<0.001$ ;  $n=39$ ). Soil moisture (SM), microbial biomass carbon (MBC), soil temperature (Ts), Aggregate Stability (Agg. Sta.), Bulk density (BD), Phospholipid fatty acid (PLFA), Cation exchange capacity (CEC)

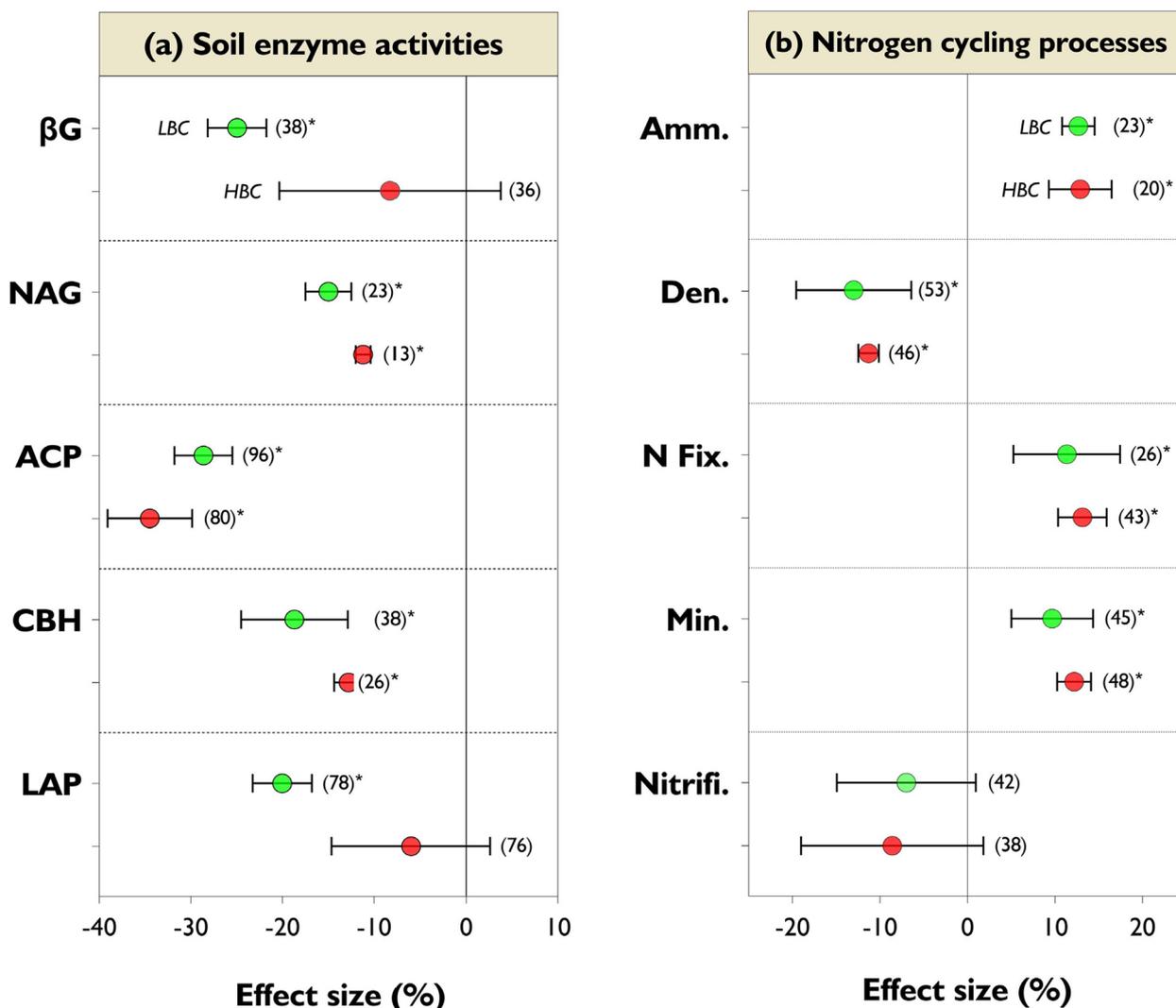
(+6.2%, CI: 4.8%–7.6%), salinity (+29%, CI: 22%–36%), CEC (+20%, CI: 15%–25%), base saturation (+21%, CI: 16%–26%), and stoichiometric ratios (C:N+20%, C:P+13%, N:P+14%) ( $p<0.05$ , Fig. 1). However, it reduced LOC (−8%, CI: −12% to −4%), NO<sub>3</sub><sup>−</sup>-N (−19%, CI: −25% to −13%), NH<sub>4</sub><sup>+</sup>-N (−23%, CI: −29% to −17%), NH<sub>3</sub>-N (−26%, CI: −33% to −19%), and NO<sub>2</sub><sup>−</sup>-N (−38%, CI: −45% to −31%) (Fig. 1;  $p<0.01$ ). In terms of biological properties, biochar increased total PLFA (+21%, CI: 15%–27%) but decreased MBC (−6%, CI: −9% to −3%) (Fig. 1;  $p<0.01$ ).

### 3.2 Soil properties influence effects of biochar application on GHG emissions

We performed a mixed-effects meta-regression analysis that identifies key factors influencing GHG emissions, including CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O (Fig. 2). Base saturation ( $r=-0.31$ ) and CEC ( $r=-0.37$ ), emerged as the most significant negative predictors of CO<sub>2</sub> emissions (Table S3). For CH<sub>4</sub> emissions, we found significant negative correlations with both soil porosity ( $r=-0.09$ ) and CEC ( $r=-0.19$ ) (Table S3). The strongest relationship was observed for N<sub>2</sub>O emissions, which showed a pronounced negative correlation with biochar application rate ( $r=-0.91$ , Table S3). While biochar application rate was the dominant factor for N<sub>2</sub>O reduction, we also detected a modest but significant negative correlation with soil properties ( $r=-0.05$ ).

### 3.3 Soil enzyme activities and nitrogen cycling processes

The results of our meta-analysis show that biochar amendment generally negatively impacts soil enzyme activities (Fig. 3). Specifically, β-glucosidase (βG) activity decreased significantly with low biochar (LBC) ( $p<0.05$ ) but not with high biochar (HBC) amendment (Fig. 3a). Similar trends were observed for other soil enzymes. NAG activity was reduced by 14% (CI: −19% to −9%) under LBC but not in HBC, compared to the control (Fig. 3a;  $p<0.05$ ). ACP activity decreased with biochar, with HBC showing a higher reduction (−34%, CI: −40% to −28%) compared to LBC (−28%, CI: −34% to −22%) (Fig. 3a;  $p<0.05$ ). Leucine aminopeptidase (LAP) also showed reductions, with LBC at −20% (CI: −26% to −14%) and HBC at −6% (CI: −12% to −1%) (Fig. 3a). In contrast, β-cellobiohydrolase (CBH) activity was higher in LBC by 18% (CI: 12%–24%) compared to HBC at 12% (CI: 6%–18%) (Fig. 3a;  $p<0.05$ ). Biochar amendment positively affected N cycling processes, significantly increasing ammonification by 12% (CI: 6%–18%) in HBC ( $p<0.05$ ). N-fixation and mineralization were enhanced by 11%–13% and 9%–12%, respectively (Fig. 3b;  $p<0.05$ ). Conversely, denitrification decreased by 10%–12% ( $p<0.05$ ), while nitrification declined in LBC (−6%, CI: −11% to −1%) and HBC (−9%, CI: −14% to −4%) (Fig. 3b).



**Fig. 3** Effects of biochar amendment on (a) soil enzyme activity and (b) nitrogen cycling processes (soil ammonification, denitrification, nitrogen fixation, nitrogen mineralization, and nitrification rates). The activity of β-glucosidase (βG); the activity of N-acetyl glucosamine (NAG), the activity of acid phosphatase (ACP), the activity of β-cellulohydrolase (CBH), the activity of Leucine aminopeptidase (LAP). Soil ammonification (Amm.), denitrification (Den.), nitrogen fixation (N Fix.), nitrogen mineralization (Min.), and nitrification (Nitrifi.). The line in red represents the control treatment. The line in green represents the control treatment. Low biochar amendment (LBC) and High biochar amendment (HBC)

### 3.4 Soil GHG emissions upon biochar amendments

#### 3.4.1 Soil CO<sub>2</sub> emissions

The average effect of biochar on soil CO<sub>2</sub> emissions varied between -14% and -51%, with a mean of 24%. Active biochar (ActBC) amendments had a greater impact on soil CO<sub>2</sub> emissions than biochar processed

at temperatures below 200 °C (Fig. 4a). The positive effect of biochar was significant only at high application rates (HBC; ≥ 40 t.ha<sup>-1</sup>) (+ 46%, CI: 38%–54%) (Fig. 4a; *p* < 0.05). The type of biochar and fertilizer treatment also influenced soil CO<sub>2</sub> emissions. BIN treatments showed significant increases (*p* < 0.05) in soil CO<sub>2</sub>

(See figure on next page.)

**Fig. 4** Effects of biochar amendment on (a) CO<sub>2</sub>, (b) CH<sub>4</sub>, and (c) N<sub>2</sub>O emission rates following biochar processing, rate, treatment type, incubation time, and the interaction between biochar addition with temperature and soil moisture (SM). Black bars represent 95% confidence intervals. The different letters in parentheses indicate the number of observations. A dashed vertical line is drawn at a mean effect size of 0. \*Significant at the 0.05 level. Biochar (BC), inorganic nitrogen (IN form, NH<sub>4</sub>NO<sub>3</sub>), biochar x nitrogen (BIN), and active-biochar (ActBC). The line in red represents the control treatment

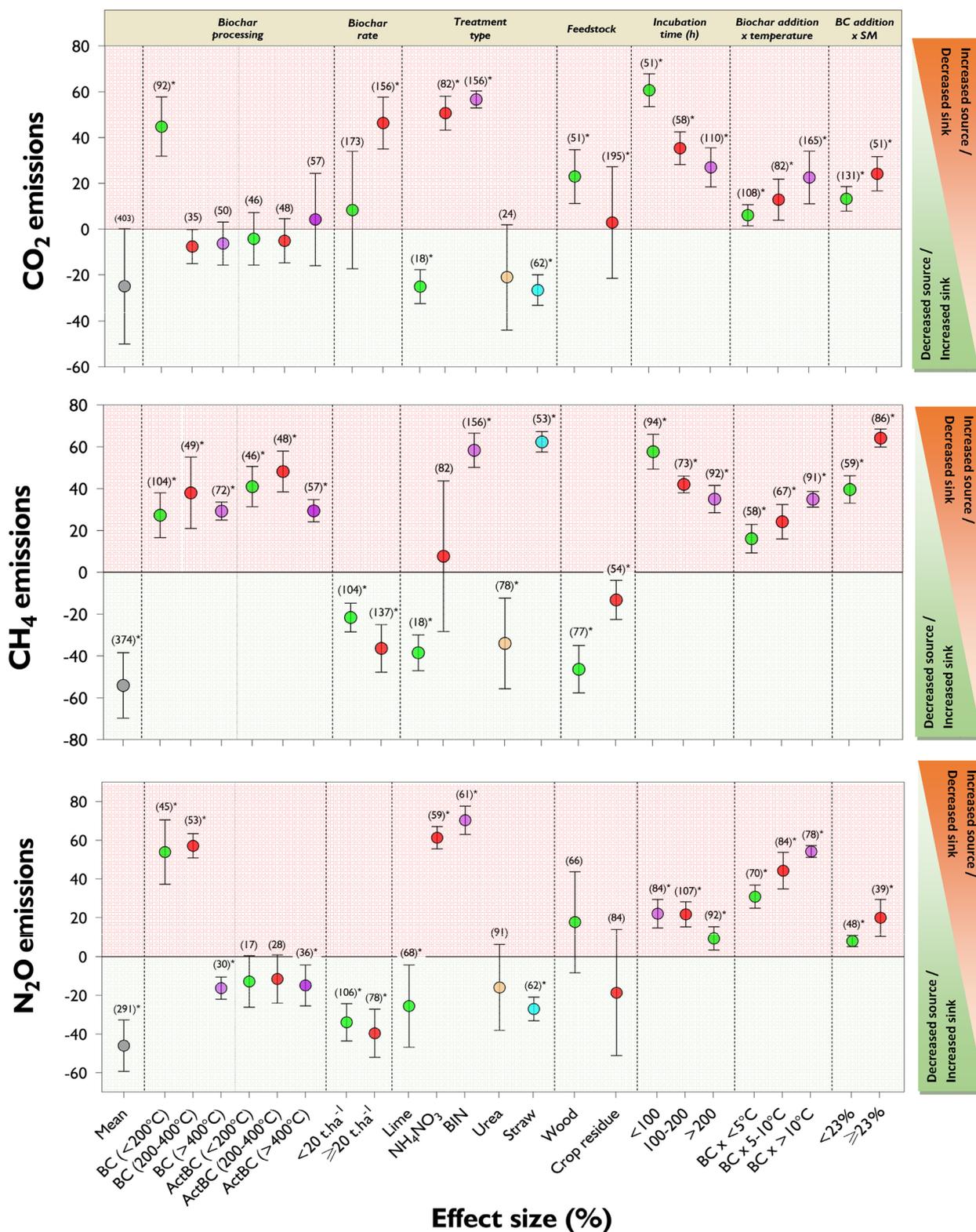


Fig. 4 (See legend on previous page.)

emissions of 56% (CI: 48%–64%) and 50% (CI: 42%–58%) compared to urea, straw, and lime amendments (Fig. 4a). Crop residue biochar reduced emissions compared to wood-derived biochar (Fig. 4a). Soil CO<sub>2</sub> emissions significantly decreased with longer biochar incubation times. At short-term amendments (100 h), emissions increased by 67% (CI: 59%–75%) (Fig. 4a) compared to medium (100–200 h) and long-term (>200 h) amendments, which increased by 35% (CI: 27%–43%) and 27% (CI: 19%–35%) (Fig. 4a;  $p < 0.05$ ), respectively. Combined effects of biochar with elevated temperature (biochar x temp.) and water availability (biochar x SM) also increased emissions. Soil CO<sub>2</sub> emissions were higher at temperatures over 10 °C and high soil moisture ( $\geq 23\%$ ) by 22% (CI: 16%–28%) and 24% (CI: 18%–30%), respectively (Fig. 4a;  $p < 0.05$ ).

### 3.4.2 Soil CH<sub>4</sub> emissions

Soil CH<sub>4</sub> emissions varied from 17% to 68% in response to biochar (Fig. 4b). Both biochar and ActBC amendments significantly increased soil CH<sub>4</sub> emissions compared to the control (Fig. 4b). Higher biochar application rates reduced soil CH<sub>4</sub> emissions by 22% (LBC, CI: –28% to –16%) and 36% (HBC, CI: –42% to –30%) (Fig. 4b;  $p < 0.05$ ). Wood-derived biochar had a greater mitigation potential (–36%, CI: –42% to –30%) than crop residue-derived biochar (–13%, CI: –19% to –7%). There was a significant decrease in soil CH<sub>4</sub> emissions over biochar incubation time (Fig. 4b;  $p < 0.05$ ). Elevated temperature and water availability increased soil CH<sub>4</sub> emissions, with water availability having a greater impact. Biochar enhances CH<sub>4</sub> uptake in aerobic soils by improving soil porosity by up to 65% (Fig. 1).

### 3.4.3 Soil N<sub>2</sub>O emissions

Biochar significantly reduced soil N<sub>2</sub>O emissions compared to controls (Fig. 4c;  $p < 0.05$ ). Processing biochar and ActBC at temperatures >400 °C decreased N<sub>2</sub>O emissions by 16% (CI: 10%–22%) and 14% (CI: 8%–20%), respectively (Fig. 4c;  $p < 0.05$ ). Biochar amendment resulted in a decrease of 39% (CI: –45% to –33%) at a LBC rate <40 t.ha<sup>–1</sup> and 33% (CI: –39% to –27%) at a HBC rate  $\geq 40$  t.ha<sup>–1</sup> (Fig. 4c;  $p < 0.05$ ). In contrast, straw (–27%, CI: –34% to –20%), lime (–25%, CI: –32% to –18%), and urea (–15%, CI: –22% to –8%) led to lower N<sub>2</sub>O emissions than the control (Fig. 4c;  $p < 0.05$  except for straw). Crop residue-derived biochar had a higher mitigation potential than wood-derived biochar (Fig. 4c). Soil N<sub>2</sub>O emissions varied by 30% to 54% with elevated temperature, while variations in water availability ranged from 8%–19% (Fig. 4c), indicating that temperature is a more significant factor influencing soil N<sub>2</sub>O emissions than water availability.

## 3.5 Biochar effects on GWP and GHGI

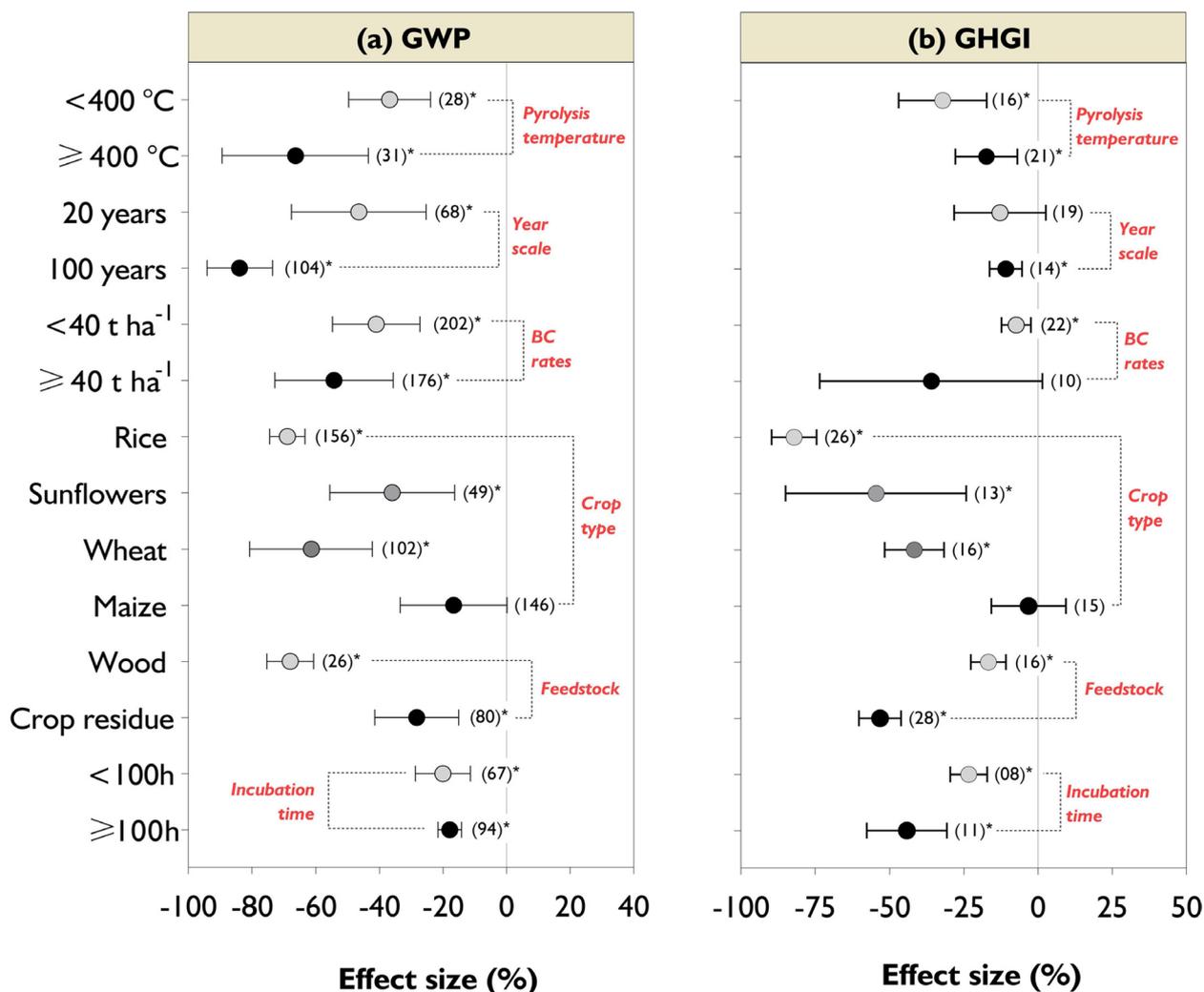
The use of biochar in agriculture can reduce GWP and GHGI (Fig. 5). To ensure comparability, GWP and GHGI values were normalized per unit of crop yield. Biochar amendment decreased GWP significantly (Fig. 5a). High pyrolysis temperatures ( $\geq 400$  °C), long-term scale (100 years), and high biochar application rates ( $\geq 40$  t.ha<sup>–1</sup>) showed GWP reductions of –66% (CI: –72% to –60%), –83% (CI: –89% to –77%), and –54% (CI: –48% to –60%) ( $p < 0.01$  for all), respectively (Fig. 5a). The 100-year timeframe is relevant for assessing long-term carbon sequestration, while the 20-year horizon reflects shorter-term climate impacts. After biochar application, the GWP of rice systems was reduced, while maize systems contributed most to GWP among the analyzed systems (Fig. 5a;  $p < 0.01$ ). Crop residue-derived biochar had a higher GWP than wood-derived biochar (Fig. 5a). Incubation time with biochar also significantly decreased GWP (Fig. 5a;  $p < 0.01$ ).

A significant difference in GHGI reduction was observed with biochar from different pyrolysis temperatures. Biochar from low temperatures (<400 °C) led to a greater GHGI reduction (–32%, CI: –38% to –26%) than that from high temperatures ( $\geq 400$  °C) (–17%, CI: –23% to –11%) (Fig. 5b;  $p < 0.01$ ). Biochar addition reduced GHGI over the short term (20 years) by 12% (CI: –18% to –6%), with no significant difference at the 100-year scale (Fig. 5b). The 20-year horizon is critical for evaluating near-term mitigation potential, while the 100-year assessment captures long-term carbon stability. High biochar (HBC) addition decreased GHGI more (–35%, CI: –41% to –29%) than low biochar (LBC) (–7%, CI: –13% to –1%;  $p < 0.01$ ). Similar to GWP, maize systems exhibited the highest GHGI, while rice systems had the lowest among the analyzed crops (Fig. 5b). On average, wood-derived biochar and crop residue-derived biochar reduced GHGI by 16% (CI: –22% to –10%) and 53% (CI: –59% to –47%), respectively (Fig. 5b;  $p < 0.01$  for all). Significant GHGI reductions occurred across all incubation periods, with the greatest reduction (–29%, CI: –35% to –23%) observed at long incubation times ( $\geq 100$  h) (Fig. 5b;  $p < 0.01$ ).

## 4 Discussion

### 4.1 Soil properties shift impact on GHG emissions

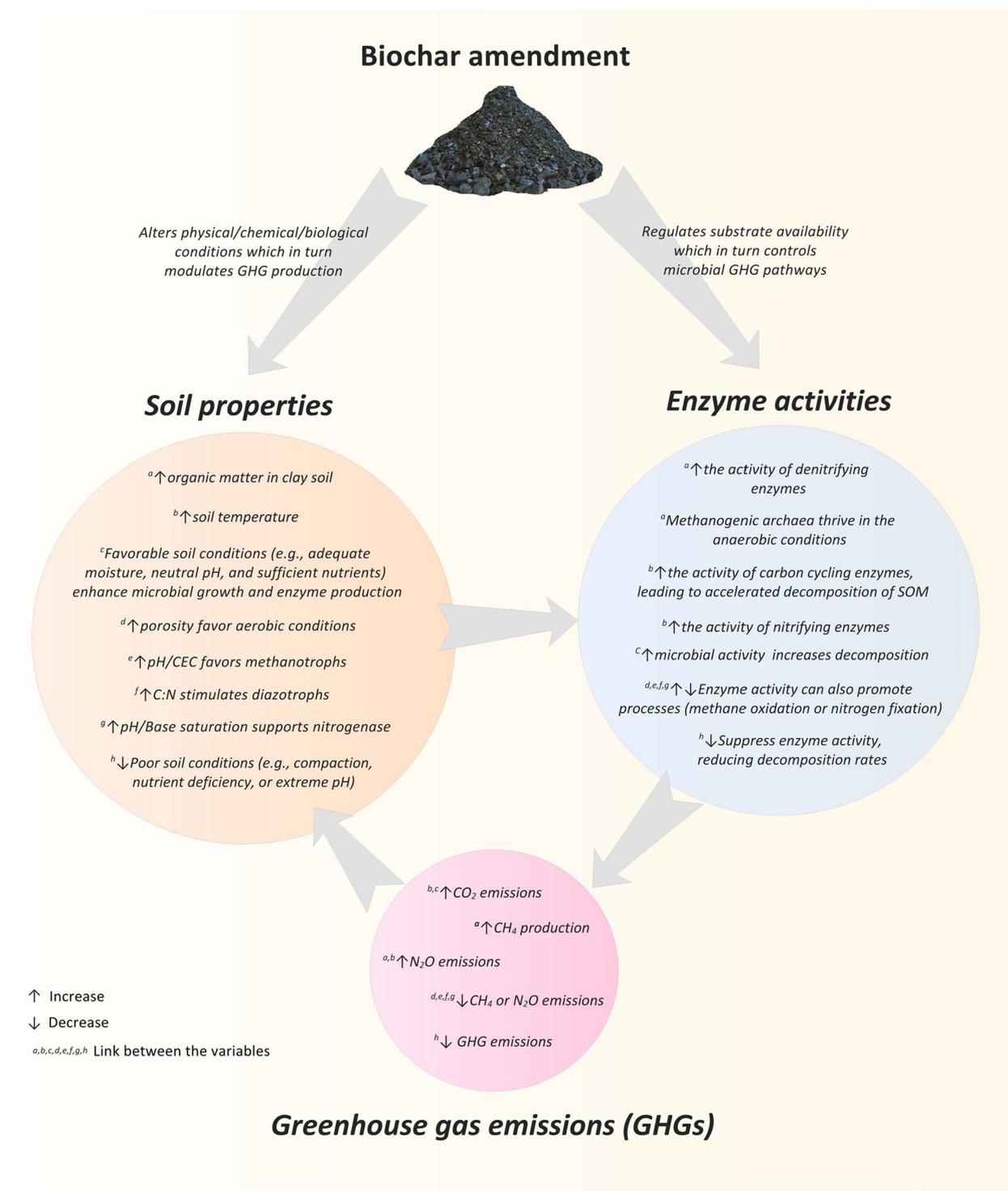
Our meta-analysis demonstrates that biochar amendment generally leads to positive effects on soil properties, thereby contributing to reductions in GHG emissions (Fig. 1). Specifically, biochar-induced enhancement in soil structure and fertility by increasing pore space creates a better habitat for beneficial organisms, including earthworms and microorganisms (Bamminger et al. 2014; Ding et al. 2016). This improvement in soil



**Fig. 5** Effect of biochar application on the global warming potential (GWP) and greenhouse gas emission intensity (GHGI) of various crops under different parameters. Numbers in parentheses indicate the numbers of observations and error bars represent the 95% confidence intervals

structure also enhances drainage, aeration, and moisture retention that, in turn, leads to reduced CO<sub>2</sub> emissions and promotes plant growth and photosynthesis (Ball 2013). Biochar’s alkaline nature can increase soil pH, a finding supported by research demonstrating significant shifts in soil pH due to biochar application (Liu et al. 2015). This increase in pH primarily occurs through the release of base cations (Ca<sup>2+</sup>, Mg<sup>2+</sup>, K<sup>+</sup>) and carbonate species from its mineral ash components (Li et al. 2013; Zhao et al. 2013). Although biochar itself does not contain pre-formed calcium carbonate (CaCO<sub>3</sub>) or magnesium carbonate (MgCO<sub>3</sub>), the available Ca<sup>2+</sup> and Mg<sup>2+</sup> can react with bicarbonate (HCO<sub>3</sub><sup>-</sup>) in the soil to form these carbonate minerals over time, particularly in alkaline conditions (Bruun et al. 2013). This process of

mineral carbonation contributes to long-term carbon sequestration by converting CO<sub>2</sub> sourced from soil respiration or atmospheric sources into stable inorganic carbonates (Buss et al. 2019). Furthermore, biochar serves as an effective carbon sink by incorporating stable aromatic carbon structures that resist microbial decomposition for centuries to millennia (Jianhua and Chen 2014). This carbon is derived from the biomass feedstock that absorbed atmospheric CO<sub>2</sub> during plant growth, effectively locking it away in the soil. Consequently, the combination of these processes, i.e., mineral carbonation and physical carbon stabilization, enhances the soil’s overall carbon storage potential while also mitigating CO<sub>2</sub> emissions (Bruun et al. 2013; Buss et al. 2019).



**Fig. 6** Relationship between GHG emissions, soil properties, and enzyme activities. The arrows indicate how the dynamics of the variables or mechanisms work, and the small letters represent their links

Biochar amendment enhances soil aggregation, as evidenced by increased aggregate stability (Ebrahimi and Or 2017), which can reduce methane (CH<sub>4</sub>) emissions

through interconnected mechanisms. The improved soil structure creates unfavorable conditions for methanogenic archaea by: (1) increasing oxygen diffusion into

previously anaerobic microsites, thereby suppressing obligate anaerobes (Keiluweit et al. 2016); (2) altering water-filled pore space distribution, which reduces the volume of strictly anoxic habitats (Lacroix et al. 2023); and (3) modifying redox gradients that control methanogenesis (Fig. 6) (Herrero et al. 2021). Simultaneously, biochar's porous structure provides favorable habitats for methanotrophic bacteria that oxidize  $\text{CH}_4$  (Reddy et al. 2014). Additionally, biochar's adsorption capacity can physically retain methane molecules, increasing their exposure to oxidative processes (Sadasivam and Reddy 2015). These mechanisms collectively shift the balance between methane production and consumption, as demonstrated by enhanced microbial methane oxidation in landfill cover soil amended with biochar.

Biochar amendment significantly alters soil microbial communities, as evidenced by changes in total PLFAs (Harter et al. 2014). While an increase in PLFAs often reflects a shift in microbial community composition, it does not necessarily indicate an overall increase in microbial biomass. Instead, biochar promotes the growth of specific microbial groups involved in nitrogen cycling, leading to a more efficient use of nitrogen and reduced  $\text{N}_2\text{O}$  emissions (Nash et al. 2021; Siles et al. 2022). This community restructuring occurs despite biochar's ability to suppress specific enzyme activity, benefiting from microbial functional redundancy and specialized niches created by biochar. In particular, biochar supports the growth of three key microbial groups: (1) denitrifying bacteria, which completely reduce  $\text{N}_2\text{O}$  to  $\text{N}_2$ , enhanced by the creation of anaerobic microsites within the biochar structure (Harter et al. 2014); (2)  $\text{N}_2$ -fixing bacteria, which benefit from the improved habitat quality provided by biochar (Ducey et al. 2015); and (3) microbial groups that compete with nitrifiers for ammonium, thereby reducing the availability of nitrogen for nitrification processes (Zhao et al. 2025). These shifts in microbial community structure contribute to more efficient nitrogen utilization and a reduction in  $\text{N}_2\text{O}$  production pathways (Zhang et al. 2021). The apparent contradiction between changes in phospholipid fatty acids (PLFAs) and potential reductions in overall microbial biomass can be clarified by recognizing that biochar selectively enhances specific functional groups within the microbial community. While the total microbial biomass may decrease or remain stable, the composition shifts toward microbes that are more efficient at nitrogen cycling, thereby reducing nitrogen losses (Sun et al. 2012). For instance, biochar creates conditions favorable for complete denitrification, characterized by low oxygen levels and high organic matter availability, which supports denitrifying bacteria (Abbas et al. 2024; Khan et al. 2022). Consequently, even within a potentially

smaller or more stable overall microbial community, the net effect is improved nitrogen retention and reduced  $\text{N}_2\text{O}$  emissions (Xu et al. 2014).

#### 4.2 Soil enzyme activity shifts the biochar impact on GHG emissions

Biochar significantly influences soil enzyme activities, primarily inhibiting  $\beta$ -glucosidase ( $\beta\text{G}$ ), N-acetylglucosaminidase (NAG),  $\beta$ -cellobiohydrolase (CBH), ACP, and leucine aminopeptidase (LAP) (Fig. 3a). This inhibition is likely due to biochar's high surface area and porous structure, which can adsorb these enzymes and reduce their catalytic efficiency (Foster et al. 2018). Consequently, the breakdown of cellulose and hemicellulose is slowed, delaying plant residue decomposition and promoting the accumulation of carbon-rich compounds in the soil (Tisserant and Cherubini 2019; Wu et al. 2013). Furthermore, the suppression of enzyme activity has significant implications for GHG emissions (Fig. 6). For instance, the inhibition of CBH activity decreases cellulose breakdown, thereby limiting substrates for methanogenesis, which leads to reduced  $\text{CH}_4$  emissions (Ding et al. 2023). Similarly, biochar's impact on ACP activity driven by its high surface area and alkaline pH further suppresses  $\text{CH}_4$  production.

The correlation between ACP activity and  $\text{CH}_4$  emissions has gained attention in agricultural practices and ecosystem management. A decrease in ACP activity may lower  $\text{CH}_4$  emissions by altering microbial dynamics and nutrient cycling in soil. Organic fertilization can modify soil microbial communities and influence greenhouse gas emissions, including methane. For instance, organic fertilizers can significantly reduce  $\text{CH}_4$  emissions compared to conventional methods due to changes in microbial dynamics and nutrient availability (Sampanpanish 2012). As ACP activity decreases, it may limit phosphorus availability, affecting microbial metabolism and the balance of  $\text{CH}_4$  production and oxidation (Kim et al. 2021). Additionally, studies on transgenic rice varieties suggest that specific genetic modifications can impact methane emissions. For example, Shi et al. (2018) found that rice plants with overexpressed genes for plant peptides reduced cumulative  $\text{CH}_4$  emissions due to altered microbial activities in the rhizosphere (Jiang et al. 2017). This reduced ACP activity may create a less favorable environment for methanogenesis (Kim et al. 2021; Zhang et al. 2015).

In addition to methane, lower  $\text{N}_2\text{O}$  emissions are also a consequence of biochar's influence. The inhibition of  $\beta\text{G}$  activity restricts the availability of labile carbon, a key substrate for denitrification, thereby reducing  $\text{N}_2\text{O}$  emissions (Zhang et al. 2023). Likewise, diminished LAP activity limits proteolysis and subsequent nitrogen

mineralization, further mitigating  $N_2O$  release (Shen et al. 2022). These findings highlight biochar's regulatory role in soil enzyme activity and its cascading effects on GHG emissions. However, it is important to note that the extent of inhibition depends on biochar properties (e.g., porosity, pH) and soil conditions, suggesting context-specific outcomes.

#### 4.3 Soil nitrogen cycling processes shift the biochar impact on GHG emissions

Biochar exerts complex, often opposing effects on soil nitrogen cycling processes and associated GHG emissions. The observed increase in total nitrogen (TN) coupled with decreases in inorganic nitrogen species such as ammonium ( $NH_4^+-N$ ) and nitrite ( $NO_2^- -N$ ) suggests a shift in nitrogen speciation within the soil following biochar amendment. This pattern likely reflects an increase in organic nitrogen pools, as biochar can enhance soil organic matter stabilization and promote the retention of nitrogen in organic forms (Oo et al. 2018).

Biochar's porous structure and adsorption capacity may protect organic nitrogen compounds from rapid mineralization, leading to the accumulation of organic N while reducing readily available inorganic N forms. The increase in  $N_2$  gas emissions could indicate enhanced denitrification completeness, where nitrous oxide ( $N_2O$ ) is further reduced to dinitrogen ( $N_2$ ), a benign end product of denitrification (Harter et al. 2014). Biochar amendments can create microsites with low oxygen conditions favorable for denitrifying bacteria, allowing them to perform complete denitrification, thus increasing  $N_2$  emissions while decreasing intermediate inorganic nitrogen species and  $N_2O$  emissions (Shan and Coleman 2020). This process contributes to nitrogen loss as gaseous  $N_2$  but reduces the emission of potent greenhouse gases like  $N_2O$ . Therefore, the seemingly contradictory trends reflect biochar's dual role in (1) stabilizing organic nitrogen in soil (Laird et al. 2010) and (2) promoting microbial processes that convert inorganic nitrogen intermediates into inert  $N_2$  gas (Egamberdieva et al. 2021). This dynamic underscores biochar's capacity to enhance nitrogen retention in organic forms while mitigating greenhouse gas emissions through improved nitrogen cycling efficiency (Wali et al. 2020).

While its alkaline properties can suppress denitrification by raising soil pH and limiting organic carbon availability for denitrifiers (Obia et al. 2015), biochar simultaneously enhances ammonification through several mechanisms. Although biochar's carbon structure is largely stable, its surface contains labile fractions (typically <5% of total C) that provide limited but strategic energy sources for microbial activity. More importantly, biochar's porous structure and high surface area (i)

protect organic nitrogen compounds from rapid decomposition while (ii) creating microhabitats that concentrate extracellular enzymes and microbes involved in nitrogen mineralization (Ahmad et al. 2021). This explains how biochar can both stabilize carbon overall (Sohi et al. 2010) while selectively enhancing specific nitrogen transformations. For nitrogen fixation, biochar improves conditions through physical protection of rhizobia, moisture retention, and provision of trace (Xiu et al. 2021), demonstrating its multifaceted role in regulating nitrogen cycling processes.

Biochar can potentially reduce  $CH_4$  emissions by enhancing soil fertility, promoting the growth of beneficial microorganisms, and suppressing methane-producing bacteria (Kammann et al. 2017). It decreases denitrification and associated  $N_2O$  emissions by creating microenvironments that favor  $N_2O$ -reducing microorganisms while inhibiting  $N_2O$ -producing microorganisms (Liu et al. 2021). The biochar-induced modification of soil physical properties hinders the activities of nitrifying microorganisms by promoting the growth of less efficient nitrifying bacteria and fungi (Blanco-Canqui 2017; Duan et al. 2019; Zhang et al. 2021). By limiting the population of ammonia-oxidizing bacteria, biochar restricts the conversion of ammonium to nitrate, thereby reducing nitrate availability for denitrification and decreasing  $N_2O$  production during nitrification (Sánchez-Monedero et al. 2018; Yao et al. 2022).

#### 4.4 Factors controlling GHG emission, GWP and GHGI following biochar amendment

The effects of biochar amendment on greenhouse gas emissions, particularly carbon dioxide ( $CO_2$ ), methane ( $CH_4$ ), and nitrous oxide ( $N_2O$ ), are intricately regulated by various soil factors including SM, microbial biomass carbon (MBC), and aggregate stability (Agg. Sta.). The interactions among these variables significantly influence the emissions of these gases in agricultural soils. In terms of  $CO_2$  emissions, biochar application can enhance soil microbial activity, leading to increased  $CO_2$  production as microbes utilize the carbon content of biochar. For instance, the addition of biochar has been shown to stimulate mineralization processes, resulting in elevated  $CO_2$  emissions from treated soils compared to controls (Hoang and Maeda 2017; Oo et al. 2018). The relationship between MBC and  $CO_2$  emissions is also evident, as higher microbial biomass often correlates with increased respiration rates and subsequent  $CO_2$  release (Hardy et al. 2019). Additionally, higher moisture contents resulting from biochar amendment favor microbial growth and can influence respiration rates, potentially leading to variations in  $CO_2$  emissions depending on the inherent moisture extremes of the soil (Ulyett et al. 2013).

Similarly, the regulation of CH<sub>4</sub> emissions by biochar is closely tied to the prevailing SM conditions and aggregate stability. Recent findings illustrate that biochar amendments may reduce CH<sub>4</sub> emissions through enhanced aeration and decreased anaerobic conditions that favor methane production (Uusitalo and Leino 2019). Soil moisture plays a critical role, since higher moisture content can promote anaerobic pathways that increase CH<sub>4</sub> emissions, but can still decrease denitrification, which is linked to N<sub>2</sub>O reduction (Kolesnikova et al. 2020). Furthermore, aggregate stability, which improves with biochar addition, is essential for maintaining soil structure, thus impacting both moisture retention and gas diffusion properties in the soil (Lan et al. 2018).

The regulation of N<sub>2</sub>O emissions by biochar is particularly notable, where both the quantity of biochar applied and its specific characteristics play a decisive role. Varying types of biochar can lead to significant differences in N<sub>2</sub>O output due to their distinct chemical properties and interactions with soil processes, such as nitrification and denitrification (Cayuela et al. 2013; Singh et al. 2010). For instance, biochars produced at high pyrolysis temperatures are associated with significantly lower N<sub>2</sub>O emissions by enhancing nitrogen retention and modifying microbial community dynamics (Lan et al. 2018). Additionally, the presence of phospholipid fatty acids (PLFAs), which indicate microbial biomass, has been identified as a significant variable influencing N<sub>2</sub>O emissions, suggesting that the microbial community structure and functional capacity play critical roles in mediating emissions in biochar-amended soils (Hardy et al. 2019).

There are several factors that influence the responses of GHG emissions to biochar amendment, including biochar processing, application rate, and incubation time (Fig. 4). The processing temperature of biochar significantly affects its physicochemical properties, such as surface area, pore size distribution, and elemental composition (Muzyka et al. 2023). It has been shown that higher processing temperatures reduce N<sub>2</sub>O emissions through NH<sub>4</sub><sup>+</sup> ion adsorption (Lan et al. 2018). Higher biochar application rates reduce emissions of GHGs, particularly of N<sub>2</sub>O and CH<sub>4</sub>, through carbon adsorption, and changes in soil microbial communities. Longer incubation times lead to decreased GHG emissions by suppressing microbial activities and decomposition processes (Jiang et al. 2021). The amendment of soils with biochar can differently affect the GHG intensity of various crops, including reductions of CO<sub>2</sub> and N<sub>2</sub>O emissions in sunflower and rice production. Maize was found to have a higher GHGI compared to other crops. This could be due to its C<sub>4</sub> photosynthetic pathway and higher nutrient demands.

The impact of biochar on greenhouse gas intensity (GHGI) varies significantly across different crops due to distinct physiological, agronomic, and soil microbial interactions (Singh et al. 2021). Rice systems exhibit the lowest GHGI with biochar application, primarily because flooded rice paddies are major sources of CH<sub>4</sub> (Han et al. 2016). Biochar mitigates this by reducing methanogenesis through the adsorption of LOC and improving redox conditions (Liu et al. 2021). It also enhances methane oxidation by increasing soil porosity, which promotes methanotroph activity (Singh et al. 2021). Additionally, biochar suppresses N<sub>2</sub>O emissions by adsorbing NH<sub>4</sub><sup>+</sup>, a critical factor in fertilized paddy soils (Huang et al. 2024). This suppression, combined with improved rice yields of around 15%–20%, further lowers GHGI by increasing greenhouse gas efficiency per unit of yield (Wang et al. 2018). Sunflower systems also benefit from biochar, showing moderate GHGI reductions. Sunflowers, with their C<sub>3</sub> physiology, generally have lower nitrogen demand compared to maize, which can reduce the risk of N<sub>2</sub>O emissions (Sun et al. 2019). Biochar stabilizes nitrogen in the soil by increasing CEC, minimizing nitrogen leaching and denitrification. It also enhances nitrogen-use efficiency, which can reduce fertilizer-derived N<sub>2</sub>O emissions (Hseu et al. 2014). In contrast, maize systems often exhibit higher GHGI despite biochar use. Maize's C<sub>4</sub> pathway and high growth rates demand more nitrogen, which intensifies N<sub>2</sub>O emissions from fertilizers (Liu et al. 2023). The addition of labile carbon from maize residues combined with biochar may temporarily boost CO<sub>2</sub> emissions. Furthermore, maize's high nitrogen uptake can limit microbial nitrogen immobilization, leaving excess NO<sub>3</sub><sup>-</sup> available for denitrification (Ndor et al. 2015). Conventional maize systems, which often rely on high synthetic nitrogen fertilizers, exacerbate N<sub>2</sub>O fluxes that biochar cannot fully offset (Benchaar et al. 2023). These differences underscore the importance of tailoring biochar applications to specific crop systems and soil conditions to maximize their benefits in reducing GHG emissions.

## 5 Study limitations

The effects of biochar on soil properties and GHG emissions vary significantly based on the type of biochar, production methods, and feedstock materials. This variability may limit the generalizability of findings. Additionally, results can be influenced by specific soil types and climatic conditions during experiments, affecting their applicability to other regions. While the study primarily focuses on biochar, it is important to note that it is often used in conjunction with other soil amendments. The interactions between these amendments are not fully explored, which may further impact biochar's

effectiveness in mitigating GHG emissions. Moreover, longer-term studies are needed to fully assess the sustainability and efficacy of biochar as a climate mitigation strategy. We acknowledge that the reliance on field studies with durations that rarely exceed 5 years introduces uncertainty regarding the long-term effects of biochar application on soil processes and greenhouse gas emissions. While our meta-analysis did not directly quantify the interactive effects of soil type and climate on biochar's impact on GHG emissions due to limitations in the available published data, it is crucial to acknowledge these factors as key contextual determinants of biochar effectiveness. Specifically, soil type (*e.g.*, sandy vs. clay, acidic vs. alkaline) and climatic conditions (*e.g.*, tropical vs. temperate, arid vs. humid) exert strong controls on soil properties and microbial processes, which in turn influence GHG emissions. Therefore, future research should prioritize studies that explicitly investigate the interactions between biochar, soil type, and climate.

## 6 Conclusions

This meta-analysis provides evidence that biochar amendments significantly reduce soil greenhouse gas (GHG) emissions through changes in soil properties, enzyme activities, and nitrogen cycling. By synthesizing data from 78 studies, we demonstrate that biochar consistently enhances soil porosity, water retention, organic carbon, and total nitrogen content while simultaneously reducing bulk density and labile organic fractions. These changes create a soil environment that is less conducive to GHG production and more favorable for carbon sequestration. Importantly, however, biochar's effectiveness varies based on its properties particularly pyrolysis temperature and application rate and the cropping system context. For instance, high-temperature biochars (> 400 °C), when applied at rates of  $\geq 40 \text{ t}\cdot\text{ha}^{-1}$  over extended periods, yield the most significant reductions in GWP, achieving up to 83% mitigation. Notably, rice-based systems benefit the most, suggesting that strategic deployment in flooded or high-emission cropping systems could maximize climate benefits. Furthermore, biochar suppresses key soil enzyme activities (*e.g.*,  $\beta$ -glucosidase, N-acetylglucosaminidase, acid phosphatase) and alters nitrogen transformation pathways. This leads to enhanced ammonification and biological nitrogen fixation while suppressing denitrification and nitrification, resulting in reduced CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions. Our findings clarify how biochar modifies the soil environment to limit GHG fluxes. These results have important implications for climate-smart agriculture: (1)

biochar selection and management should be tailored to crop type, soil conditions, and emission profiles; (2) long-term, high-rate applications of high-temperature biochar offer the greatest mitigation potential; and (3) future research should integrate biochar with other sustainable practices and explore its long-term impacts on soil health and productivity.

## Supplementary Information

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Supplementary Material 1.

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## Authors' contributions

The conception was conceived by Mbezele Junior Yannick Ngaba, who coordinated the research and designed the study along with Bin Hu and Heinz Rennenberg. They also took care of the initial title and abstract screening. Mbezele Junior Yannick Ngaba, Abubakari Said Mgelwa, Muhammed Mustapha Ibrahim, Bin Hu, and Heinz Rennenberg conducted the full-text screening. Mbezele Junior Yannick Ngaba performed data extraction. The writing of the paper was completed by Mbezele Junior Yannick Ngaba, Abubakari Said Mgelwa, Muhammed Mustapha Ibrahim, Bin Hu, and Heinz Rennenberg.

## Data Availability

The datasets used or analysed in this study are available from the corresponding author on reasonable request.

## Declarations

### Competing interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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