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Does biochar accelerate the mitigation of greenhouse gaseous emissions from agricultural soil? - A global meta-analysis

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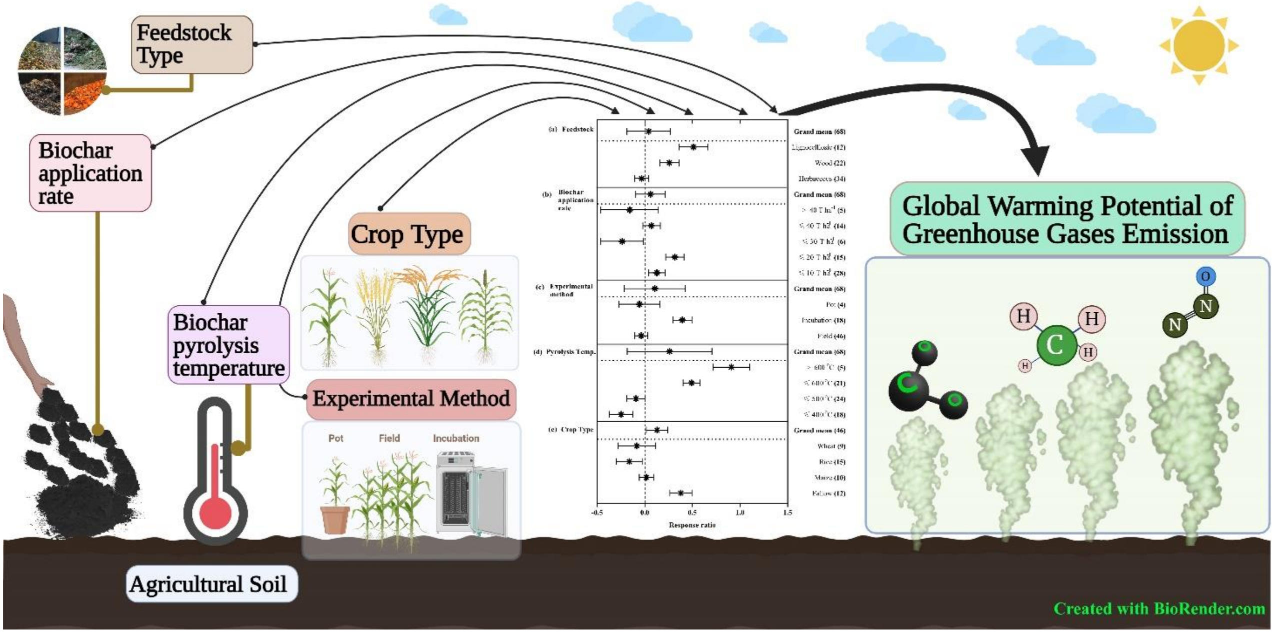
Graphical Abstract



Abstract

Greenhouse gaseous (GHGs) emissions from cropland soils are one of the major contributors to global warming; however, the extent and pattern of these climatic breakdowns are typically determined by the management practices in-place. The use of biochar on cropland soils holds a great promise for increasing the overall crop productivity. However, biochar applications to agricultural soil has grown in popularity as a strategy to off-set the negative feedback associated with agriculture GHGs emissions i.e., CO_2 (carbon dioxide), CH_4 (methane), and N_2O (nitrous oxide). Despite increased efforts to uncover biochar's potential for mitigating the farmland GHG effects, there has been little synthesis of how different types of biochar influence cropland soil GHG fluxes under varied experimental conditions. Here, we presented a meta-analysis of biochar-GHG emissions interactions across global cropland soil, with field experiments showing the strongest GHG mitigation potential i.e., CO_2 ($RR = -0.108$), N_2O ($RR = 0.11$), and CH_4 ($RR = -0.399$). The biochar pyrolysis temperature, feedstock, C: N ratio, and pH were also found to be important factors influencing GHGs emissions. A prominent reduction in N_2O ($RR = -0.13$) and CH_4 ($RR = -1.035$) emissions was observed in neutral soils (pH = 6.6-7.3), whereas acidic soils ($pH \leq 6.5$) accounted for the strongest mitigation effect on CO_2 ($RR = 0.12$) emissions. We also discovered that a biochar application rate of $\leqslant 30$ t ha^{-1} was best-suited for mitigating GHGs emissions while maintaining optimum crop yield. According to our meta-analysis, maize crop receiving biochar amendment showed a significant mitigation potential for CO_2, N_2O, and CH_4 emission. On the other hand, the use of biochar had shown significant impact on the global warming potential (GWP) of total GHGs emissions. The current data synthesis takes the lead in analyzing emissions status and mitigation potential for three of the most common GHGs from cropland soils and demonstrates that biochar application can significantly reduce the emissions budget from agriculture.

Keywords: biochar, meta-analysis, GHGs emissions, mitigation, feedstock, crop type

1. Introduction

Over the last few decades, increased fossil fuels burning together with pervasive deforestation has altered the atmospheric balance of greenhouse gases (Shukla et al., 2019), and as a result of these changes, anomalous shifts in global climate are being recorded across ecosystems (Malhi et al., 2020; Raven and Wagner, 2021). Altogether, CO_2 (carbon dioxide), CH_4 (methane), and N_2O (nitrous oxide) account for ~90% of anthropogenic global warming (Forster et al., 2007). With the scientific consensus that global warming is closely linked to changes in greenhouse gas (GHGs) emissions (Zhang et al., 2012; Shakoor et al., 2018), the future outlook seems quite bleak, as most climate models project an increase in average global temperature ranging from 1.1 to 6.4 °C over the next 100 years (IPCC, 2007). As the world becomes more vulnerable to climate related natural disasters and human health concerns, GHGs emissions across ecosystems have become one of the pressing research issues in ecology (Le Quéré et al., 2016; Qin et al., 2016).

The hovering riddle of food-security has prompted the global agriculture systems to practice intensive farming in order to meet the rising food demand of an estimated 9-billion people by 2050 (Bahar et al., 2020; Shakoor et al., 2020b). Farmers are currently inclined to use more synthetic fertilizers and pesticides to improve crop yields; however, these practices are becoming increasingly unsustainable due to the significant amount of GHG emissions into the atmosphere. As for now, nearly 11% of anthropogenic GHG emissions into the atmosphere are sourced from diverse agriculture practices, which can exacerbate the issue of climate change while putting the sustainability of agriculture sector at stake (Smith et al., 2007; IPCC, 2019).

Of various causes of global warming, perhaps none has received as much attention in the context of biosphere and ecosystem stability as CO_2. It is the primary GHG that has consistently been stocked up into the atmosphere, from 280 ppm of pre-industrial era to 415 ppm in recent times, primarily due to the unprecedented anthropogenic activities (Wu et al., 2020). More recently, N_2O emission has been identified as an imposing challenge because of its greater warming potential (298 times more than that of CO_2), although atmospheric N_2O concentration is relatively minor (Charles et al., 2017; Davidson, 2009). Most importantly, agricultural soils contribute up to 78% of the total anthropogenic N_2O emission (Mbow et al., 2017; Shakoor et al., 2021b), contributing to ~ 6% of the global greenhouse budget (Smith et al., 2007). In general, agricultural soils received a large amount of reactive N fertilizers, which can result in unintended feedback responses, such as N_2O emission ranging from 4.5 to 6.0 Tg yr^{-1} (Charles et al., 2017). Indeed, these N transformations have far-reaching environmental consequences, as N_2O is an active contributor to stratospheric ozone depletion (Portmann et al., 2012). Globally, atmospheric CH_4 concentration is hovering around 1875 ppb, more than 2.5 times higher than pre-industrial era (Dlugokencky, 2020). Among various sources, croplands account for the highest anthropogenic CH_4 emission source, contributing approximately 50% of the total CH_4 emission (Shakoor et al., 2020a; Wang et al., 2016; Xu et al., 2016). The application of chemical fertilizers, particularly N, directly or indirectly contributes to CH_4 emissions from croplands (Wang et al., 2020).

At present, crop yield uncertainties and rising GHGs emissions have marred the overall productive capacity of agriculture systems, putting future food security targets in jeopardy. Indeed, this peculiar situation emphasizes the importance of transitioning from modern intensive farming to more sustainable agricultural management, which can boost crop productivity while reducing GHGs emissions. Biochar (a C-rich charcoal material) is produced by dry carbonization process, either under the complete or partial absence of O_2, at high temperatures ranging from 300 to 1000 °C (Dahlawi et al., 2018). Globally, biochar has attracted considerable attention as a versatile organic amendment with significant potential for mitigating the global warming effects (Ashiq et al., 2020), increasing crop productivity (Lehmann and Joseph, 2015), and C-sequestration (Wang et al., 2016). The availability of wide-ranging feed-stock materials, as well as the pyrolysis temperature conditions, can produce biochar of varied physical and structural attributes, including but not limited to mechanical strength, porosity, surface area, particle size, and density and structural complexity (Lehmann and Joseph, 2015; Oni et al., 2019; Sarfraz et al., 2017). Biochar application can replenish key soil nutrients in low fertility soils due to its unique surface charge density, and the predominant negative charged surfaces of biochar also promote cation adsorption (Kongthod et al., 2015; Lou et al., 2016). Since the sources and sinks of three potent GHGs (CO_2, N_2O, and CH_4) constitute major components of the C budget across ecosystems, including biochar as a soil amendment, is critical, as it can sequester C and, more importantly, prime the soil to negate anthropogenic climate warming emissions (Montanarella and Lugato, 2013). According to estimates, biochar produced from a 2.2 Gt of feedstock-material can remove 0.49 Gt C from the atmosphere each year, implying greater merits for its use as a key climate change mitigation strategy (Woolf et al., 2010).

Taken together, previous studies of GHGs emissions response to biochar application show a variable response, such as decrease in emissions (Van Zwieten et al., 2010; Wang et al., 2012; Zhang et al., 2016), increase in emissions (Song et al., 2016; Wang et al., 2018; Wu et al., 2014) and even neutral effects (Abagandura et al., 2019; Li et al., 2021) on cropland GHGs emissions. Plausible reasons for such differences involve the use of different feed-stock materials, crop types and intensities, and experimental conditions, all of which could impact relevant soil biochemical functions mediating cropland emissions rates.

In recent decades, several studies have concentrated on cropland CO_2 emission response effect to biochar application, where an increase in CO_2 was mediated by labile C constituents of applied biochar (Zimmerman et al., 2011), and decrease has been associated with suppression of some key enzymes activity (Case et al., 2014).

Given that biochar application has been shown to improve soil conditioning attributes, such as pH, porosity, nutrient transformation, and these modified conditions may alter nitrifier and denitrifiers activity with some feed-back effect, either reduction and/ or increase, on N_2O emission flux (Cayuela et al., 2014). A valuable insight into the extent to which cropland soil releases CH_4 emissions can be attributed directly to the biochar-driven positive and negative effects on methanotrophs activity (Feng et al., 2012; Spokas, 2013). Indeed, other factors such as pyrolysis temperature, feedstock type, application rate, experimental method as well as the duration of the study could affect the overall response effect of GHGs emission to biochar amendment (Mukome et al., 2013).

Meta-analysis is a useful method to quantitatively synthesize, analyze and summarize the final results of various research studies (Freeman et al., 1986; Ren et al., 2017). While most of the biochar related meta-analyses have so far focused on the individual GHG response effect, which were either restricted to soil incubation studies or limited field scale extrapolation. To date, there is of yet no large-scale biochar data synthesis from which to draw conclusions about biochar impact on emission behavior of three potent GHGs (i.e., CO_2, N_2O, CH_4) from cropland soils.

To bridge this knowledge gap, we conducted a comprehensive global meta-analysis to determine what empirical evidence is currently available to justify the use of biochar as a climate warming mitigation amendment in cropland soil. The main objectives of this meta-analysis were: (a) to build up baseline knowledge and mechanisms of how biochar application affects greenhouse gas emissions from cropland soil, (b) to investigate the effect of different biochar properties (i.e. biochar feedstock, pyrolysis temperature, experimental method, rate of biochar application) and soil-plant system attributes (i.e. crop type and duration, soil texture, soil C: N ratio, pH and study regions) on GHGs emissions response of cropland soil.

2. Materials and methods

2.1.Literature search and data collection

The search keywords 'biochar' OR 'charcoal' OR 'char' and 'GHGs' 'carbon dioxide' OR 'CO_2' 'AND 'nitrous oxide' OR 'N_2O' AND 'methane' OR 'CH_4' were used to search the peer-reviewed articles. To fulfill the requirements of the objectives of this study, a total of thousand (1000) research articles of peer-reviewed journals were collected that reported GHGs emissions after biochar amendment to the agricultural soils in the search engines of Google Scholar, Scopus, and Web of Science, to a cut-off date of 29th February 2020. We selected peer-reviewed publications using the following criteria; a) experiments with at least one control and treatment comparison and also calculated CO_2, N_2O, and/or CH_4 emission, b) physicochemical characteristics of biochar for example pyrolysis temperature, feedstock type, C: N ratio, and pH, c) clearly described experimental method, and d) physicochemical properties of soil. For this meta-analysis study, a total of 45 peer-reviewed research articles with 297 observations (multiple treatments included within individual research articles) across 6 continents published between 2010 to 2020 were selected (Table 1).

Most publications reported emission data in tables that could be transferred into the dataset directly. Emission data presented in figures, Plot Digitizer (version 2.6.6) software was run to extract the final data. From each research publication, we extracted the cumulative values (kg ha^{-1}) of all three GHGs emissions in the dataset. We also collected the mean values, standard deviations, and sample sizes from treatment and control for each research study. If research articles only presented standard errors in the publication, the corresponding standard deviations were calculated from standard errors.

Soil data compilation

Soil data were categorized into different groups based on by following the USDA, (1999):

1) Soil texture

a) Fine (silt clay, clay, sandy clay),

b) Medium (loam, clay loam, silt, silty clay loam, silt loam), and

c) Coarse (sandy clay loam, sandy loam, loamy sand).

2) Soil pH

a) $\leqslant$ 6.5 (Acidic), b) 6.6-7.3 (Neutral), and c) > 7.3(Alkaline).

3) Soil C: N ratio

a) $\leqslant$ 10, and b) > 10.

Biochar data compilation

1) Biochar feedstock

We used the same grouping method for biochar feedstock that Cayuela et al. (2014) had used.

a) Herbaceous (straws, bamboo, green-waste), b) Biosolids (sewage sludge from treatment plants), c) Wood (willow, pine, oak, sycamore, wood mixtures), d) Manure (from pig, poultry, cattle), and e) Lignocellulosic waste (rice husk, nuts shells).

2) Pyrolysis temperature (^{o}C)

a) $\leqslant$ 400, b) $\leqslant$ 500, c) $\leqslant$ 600, and d) > 600.

3) Biochar application rate (T ha^{-1})

a) $\leqslant$ 10, b) $\leqslant$ 20, c) $\leqslant$ 30, d) $\leqslant$ 40, and e) > 40.

4) Biochar pH

a) ⩽ 6.5 (Acidic), b) 6.6-7.3 (Neutral), and c) > 7.3 (Alkaline).

5) Biochar C: N ratio

a) ⩽ 50, b) ⩽ 150, c) ⩽ 300, and d) > 300.

Other auxiliary variables

1) Experimental method

a) Pot, b) Field, and c) incubation.

2) Crop type and duration were also subdivided into different categories.

3) Study regions were subdivided as follow:

a) Asia, b) Africa, c) Europe, d) Australia, and e) America (both south and north America).

2.2.Meta-analysis

For this study, natural log response ratio (ln*RR*) was carried out to calculate effect size (Hedges et al., 1999) using the following equation:

$$\ln RR = \ln(\overline{xt}/\overline{xc}) = \ln(\overline{xt}) - \ln(\overline{xc}) \qquad (1)$$

Where the subscript of $\overline{xt}$ and $\overline{xc}$ shows the mean value of biochar treatment and control, respectively. If the lnRR value is zero, ln*RR* > 1, and ln*RR* < 1, its mean that biochar treatment had no, positive and negative effects on GHGs emissions, respectively. We used a random-effects meta-analysis model to examine our dataset as early explained by researchers (Michael et al., 2009; Skinner et al., 2014). METAWIN 2.1 (Rosenberg et al., 2000) software was used to analyze the mean effect sizes of the dataset and 95% bootstrapped confidence intervals (CIs) were generated using 4999 iterations. The results were considered significant if the 95% CI of cumulative CO_2, N_2O, and CH_4 emissions did not overlap with zero and the randomization tests resulted $p < 0.05$. The total heterogeneity (Q_t) were also calculated using METAWIN 2.1. The relationship between the variables is significant if $p < 0.05$.

3. Results

3.1. Comparative effects of biochar feedstocks and pyrolysis temperature on GHGs emissions

Figure 1 shows the overall effect sizes (response ratio (RR)) of biochar feedstock and pyrolysis temperature on CO_2, N_2O and CH_4 emissions from agricultural soils. From the 297 observations, 205 were selected for both feedstock and pyrolysis temperature. The observations contribution of biochar feedstock and pyrolysis temperature were herbaceous (n=103), biosolids (n=3), wood (n=67), manure (n=12), lignocellulosic (n=20) and ⩽ 400 °C (n=67), ⩽ 500 °C (n=66), ⩽ 600 °C (n=51), > 600 °C (n=21), respectively.

On average , biochar feedstock and pyrolysis temperature significantly increased the CO_2 emissions from soils ($RR = 0.158, 95\%\ \mathrm{CI} = -0.104, 0.421$), and ($RR = 0.11, 95\%\ \mathrm{CI} = -0.102, 0.322$), respectively (Figure 1 (I)). Of the five most commonly used feedstocks, lignocellulosic derived biochar was the most significant contributor to higher CO_2 emissions (RR = 0.497, 95% CI = 0.426, 0.568). Meanwhile, herbaceous derived biochar did not alter the pattern of CO_2 emissions (RR = -0.015, 95% CI = -0.039, 0.009). Most importantly, manure derived biochar had shown the greatest mitigation potential for CO_2 emissions (RR = -0.264, 95% CI = -0.449, -0.079). (Figure 1 (Ia)).

Across four pyrolysis temperature conditions, significantly greater CO_2 emissions were recorded at ⩽ 600 °C ($RR = 0.439, 95\%\ \mathrm{CI} = 0.389, 0.488$) followed by at ⩽ 500 °C (RR = 0.105, 95% CI = 0.07,0.14). In contrast, lowering of pyrolysis temperature to ⩽ 400 °C resulted in significant reduction in CO_2 emission ($RR = -0.046\ 95\%\ \mathrm{CI} =$

$-0.092, -0.001$). However, biochar produced at > 600 °C was marginally more responsive in mitigating the cropland CO_2 emissions ($RR = -0.061, 95\%\ \mathrm{CI} = -0.14, 0.018$) (Figure 1 (Ib)). Total heterogeneity (Q_t) of effect sizes for feedstocks and pyrolysis temperature ranged between 350.313 and 230.608, respectively, at $p < 0.001$ (Table S1).

In general, the effect size of biochar feedstock on N_2O emissions was significantly positive (*RR* = 0.122, 95% CI = -0.146, 0.39); however, the overall response effect of pyrolysis temperature did not show significant differences for N_2O emissions (*RR* = 0.002, 95% CI = -0.168, 0.171) (Figure 1 (II)). Among investigated feedstocks, manure derived biochar (*RR* = 1.082, 95% CI = 0.808, 1.357) was the leading contributor of N_2O emission from cropland soil. However, with biosolids derive biochar, cropland soils recorded a significant decrease ($RR = -0.298, 95\%\ \mathrm{CI} = -0.622, 0.026$) in N_2O emission (Figure 1 (IIa)). Similar decreasing pattern was also noticed with lignocellulosic derived biochar amendment (*RR* = -0.18, 95% CI = -0.348, -0.011).

Considering all pyrolysis temperature, biochar produced at ⩽ 400 °C ($RR = -0.13, 95\%\ \mathrm{CI} = -0.212, -0.048$) tended to decrease N_2O emissions from agricultural soils (Figure 1 (IIb)); however, biochar produced at ⩽ 500 °C demonstrated the greatest response effect (RR = -0.176,95% CI = -0.249,-0.103) for crop land N_2O mitigation. The Q_t values for feedstock and pyrolysis temperature were 66.869 and 37.339, respectively (Table S1; $p < 0.001$).

The analysis of all biochar feedstocks studies revealed an overall strong CH_4 mitigation potential in cropland soils (*RR* = -0.397, 95% CI = -0.896, 0.102) (Figure 1 (III)). Individually, each feedstock types showed a significant reduction in CH_4 emission except manure derived biochar. The maximum reduction was detected under wood derived biochar (*RR* = -1.198, 95% CI = -1.273, -1.122). Similarly, herbaceous (*RR* = -0.263, 95% CI = -0.31, -0.215) and biosolids (*RR* = -0.544, 95% CI = -1.54, 0.452) derived biochar also reduced CH_4 from agricultural soils (Figure 1 (IIIa)). The overall effect size of pyrolysis temperature on CH_4 emissions were significantly positive (*RR* = 0.254, 95% CI = -0.108, 0.615); however, the effect sizes of all subdivision were close to zero expect ⩽ 600 °C means pyrolysis temperature at ⩽ 600 °C (*RR* = 1.048, 95% CI = 0.921, 1.175) significantly increased the CH_4 emissions (Figure 1 (IIIb)). The total heterogeneity was 1178.006 and 266.960 for biochar feedstock and pyrolysis temperature, respectively (Table S1; $p < 0.001$).

3.2. Effects of experimental conditions and soil types on GHGs emissions

For CO_2 and N_2O emissions, 201 and 153 observations were selected for given experimental conditions (field n=134, incubation n=59 and pot n=8) and soil types (coarse n=81, medium n=54 and fine n=18), respectively. In the case of CH_4 emissions, 201 (field n=134, incubation n=59, and pot n=8) and 135 (coarse n=81 and medium n=54) observations were sorted out to comply with given experimental conditions and soil types, respectively. According to our meta-analysis results, both experimental conditions and soil types appeared to have significantly enhanced the CO_2 emissions from agricultural soils. Overall, soil types (*RR* = 0.555, 95% CI = -0.304, 1.414) was ascribed for greater effect size than experimental condition (*RR* = 0.197, 95% CI = -0.244, 0.638), when compared for CO_2 emissions (Figure 2 (I)).

Across experimental condition, Incubation (*RR* = 0.486, 95% CI = 0.435, 0.537) and pot (*RR* = 0.215, 95% CI = 0.101, 0.328) experiments were the leading contributors of higher CO_2 emission, while field experiments (*RR* = -0.108, 95% CI = -0.133, -0.084) recorded significant reduction of CO_2 emissions from soils (Figure 2 (Ia)). When we considered biochar application across different soil types, fine textured soils had the greatest CO_2 emissions (*RR* = 1.4, 95% CI = 1.358, 1.442) (Figure 2 (Ib)). Both experimental conditions and soil types encountered the total heterogeneity ranging between 436.387 and 2608.904, respectively (Table S1; $p < 0.001$).

The use of biochar across different experimental conditions (*RR* = 0.384, 95% CI = 0.085, 0.683) and soil types (*RR* = 0.124, 95% CI = -0.252, 0.501) led to significant overall effect and increased N_2O emissions from agricultural soils (Figure 2 (II)). Biochar application in pot experiments prompted for higher N_2O when compared with field and incubation experiments (Figure 2 (IIa)). The meta-data distribution across different soil types revealed that fine and medium soils receiving biochar had the highest N_2O emissions. While, in contrast, biochar application to coarse soils demonstrated a significant reduction in N_2O emissions (Figure 2 (IIb). The Q_t values were 33.256 and 96.975 for experimental conditions and soil types, respectively (Table S1; $p < 0.001$).

On average, experimental conditions (*RR* = -0.081, 95% CI = -0.488, 0.327) and soil types (*RR* = -0.56, 95% CI = -1.726, 0.606) showed significant CH_4 mitigation potential from croplands (Figure 2 (III)). Under experimental conditions, biochar accounted for the highest repression in CH_4 emission in field experiments (*RR* = -0.399, 95% CI = -0.435, 0.364). The pot experiment did not show any effect on CH_4 emissions, as evident by its close zero alignment. Moreover, biochar-related soil incubations showed marginal increase in CH_4 emissions (Figure 2 (IIIa)). Following biochar applications, no significant effect was detected on CH_4 emissions from medium texture soils on CH_4 emissions (*RR* = 0.036, 95% CI = -0.073, 0.146), although coarse texture soils (*RR* = -1.154, 95% CI = -1.205, -1.103) contributed to significant mitigation of CH_4 emissions from the agricultural soils (Figure 2 (IIIb)). Total heterogeneities suggest that both experimental conditions and soil types had a significant repression effect on CH_4 emissions (Table S1).

3.3. *Effect of biochar application rate on GHGs emissions and crop yield*

Data on GHGs emissions and crop yields in relation to biochar application rates were extracted from a total of 202 and 112 observations, respectively (Figure 3). On average, biochar applications significantly stimulated CO_2 emissions (*RR* = 0.185, 95% CI = 0.098, 0.272), decreased CH_4 emissions (*RR* = -0.186, 95% CI = -0.551, 0.179), and did not show any effect on N_2O emissions (*RR* = 0.008, 95% CI = -0.139, 0.156) (Figure 3 (I, II, III)).

For CO_2 emissions, all biochar application rates significantly increased CO_2 emissions; however, biochar applied at ⩽ 20 (T ha^{-1}) resulted in the highest CO_2 emissions (*RR* = 0.308, 95% CI = 0.268, 0.347), followed by biochar applied at ⩽ 10 (T ha^{-1}) (*RR* = 0.284, 95% CI = 0.249, 0.319) (Figure 3 (I).

For N_2O emissions, biochar application rates of ⩽ 10 (T ha^{-1}), ⩽ 20 (T ha^{-1}) and > 40 (T ha^{-1}) significantly enhanced N_2O emissions, whereas biochar rates of ⩽ 30 (T ha^{-1}) (*RR* = -0.256, 95% CI = -0.384, -0.128) and ⩽ 40 (T ha^{-1}) (*RR* = -0.073, 95% CI = -0.197, 0.05) biochar rates showed strong mitigating effect in N_2O emissions (Figure 3 (II).

In case of CH_4 emissions, most biochar application rates, except < 20 and < 40 T ha^{-1}, demonstrated a significant reduction in CH_4 emissions from croplands (Figure 3 (III). Most importantly, higher biochar application rate (e.g., >40 T ha^{-1}) showed the greatest potential for CH_4 mitigation from cropland soil (*RR* = -0.704, 95% CI = -0.883, -0.524) (Figure 3 (III). Across yield data (n= 112), we detected an overall positive response effect of biochar application rate on crop yield (Figure 3 (IV). Specifically, biochar application, irrespective of its application rate, significantly promoted the gains in crop yield, which was indeed further supported by total heterogeneities observed in our results (Table S1).

3.4. *Effects of biochar and soil pH on cropland GHGs emissions*

According to our meta-analysis, the overall reduction in GHGs emissions in biochar amended soil is strongly tied in with pH of added biochar and experimental soil (Figure 4). For

CO_2, all biochar pH observations (n=200) showed significant increase in CO_2 emissions (*RR* = 0.233, 95% CI = 0.138, 0.327), although differences were not significant between these different pH biochar (Figure 4 (Ia). In case of soil pH (n=191), neutral pH soil indicated the strongest effect size of biochar amendment on CO_2 emission (*RR* = 1.555, 95% CI = 1.505, 1.604), when compared with the rest of soil pH conditions (Figure 4 (Ib). The Q_t values of biochar and soil pH were (11.490; $p < 0.003$) and (2467.65; $p < 0.001$, respectively (Table S1).

Following the strong overall effect of biochar pH (*RR* = -0.073, 95% CI = -0.268, 0.123) on N_2O mitigation from cropland soils, soil pH effect (*RR* = 0.146, 95% CI = -0.141, 0.433) contradicts N_2O mitigation by significantly increasing the N_2O emissions from agricultural soils (Figure 4 (II).

When comparing acidic biochar application, it showed the strongest mitigation against N_2O emissions (*RR* = -0.262, 95% CI = -0.463, -0.06), while biochar of neutral pH promoted N_2O emissions from cropland soils (*RR* = 0.113, 95% CI = -0.297, 0.523) (Figure 4 (IIa). For different pH soils, biochar application to acidic soils generally increased N_2O emissions; however, the emission effect size was substantially larger when biochar was amended to alkaline soils (Figure 4 (IIb). Conversely, biochar applications to neutral soils showed significant potential for N_2O mitigation (*RR* = -0.13, 95% CI = -0.233, -0.027).

On average, pH of amended biochar revealed an intensifying effect on CH_4 emissions (*RR* = 0.243, 95% CI = -0.068, 0.555), while soil pH promoted the CH_4 mitigation following biochar application (*RR* = -0.428, 95% CI = -1.067, 0.21) (Figure 4 (III). Across biochar pH related observations, the maximum CH_4 emissions were recorded in acidic biochar (*RR* = 2.503, 95% CI = 0.967, 4.038) (Figure 4 (IIIa). Under soil pH conditions, biochar application showed varied but significant mitigation of CH_4 from cropland soils. Altogether, the maximum reduction in soil CH_4 emission with biochar amendment was observed in neutral soils (*RR* = -1.035, 95% CI = -1.097, -0.973) (Figure 4 (IIIb).

3.5.Effects of biochar and soil C: N ratio on cropland GHGs emissions

The response effect of GHGs emissions to biochar application also changed with biochar and soil C: N ratios (Figure 5). Biochar of varying C: N ratios significantly increased CO_2 emissions, with the exception of biochar with C: N ratio ≤ 50, which demonstrated greater mitigation potential for CO_2 from cropland soils (Figure 5 (Ia)). Nevertheless, adding biochar with a C: N of > 300 led to the highest CO_2 emissions from cropland soils. Regardless of soil C: N, biochar application caused the significant increase in CO_2 emissions, although the effect size was relatively larger in soil C: N ratio > 10 (Figure 5 (Ib)).

In terms of N_2O emissions, biochar with higher C: N (> 300 and ≤ 300) significantly reduced the cropland N_2O emissions, while biochar with lower C: N ratio (≤ 50 and ≤ 150) had shown considerable increase in N_2O emissions from cropland soils (Figure 5 (IIa)). Similarly, biochar application to soil with a given C: N ratio greatly contributed to higher N_2O emissions across cropland soils Figure 5 (IIb)).

When biochar was amended with C: N > 300 and ≤ 50, the response effect of CH_4 to biochar C: N ratio demonstrated greater CH_4 mitigation from cropland soil. On the contrary, biochar with C: N ratio (≤ 300 and ≤ 150) resulted in a marginally significant increase of CH_4 emission (Figure 5 (IIIa)). Furthermore, the CH_4 response effect to biochar application in soil with ≤ 10 C: N displayed significant CH_4 mitigation potential across cropland soil (Figure 5 (IIIb)).

3.6. Effects of crop type and duration on GHGs emissions

Figure 6 shows the overall effect sizes of crop type (n=136) and duration (n=201) in biochar amendment soils. In this meta-analysis, four type of crops (rice n=48, maize n=24, wheat n=18 and vegetables n=16) were sorted out along with fallow soils (n=30). Among all the observations, crop duration (days) were divided into ⩽ 320 days (n=110), 321-725 days (n=61) and > 725 days (n=30).

Overall effect sizes of crop types that received biochar amendment did not show any significant effect on CO_2 (*RR* = -0.003, 95% CI = -0.465, 0.459) (Figure 6 (Ia)) and N_2O emissions (*RR* = 0.021, 95% CI = -0.187, 0.228) (Figure 6 (IIa)); however, significant mitigation effects were observed in CH_4 emissions (*RR* = -0.473, 95% CI = -0.864, -0.082) (Figure 6 (IIIa)). The responses of CO_2 emissions to vegetables, wheat, maize, and rice crops after biochar application offered varied sink capacity for CO_2 (Figure 6 (Ia)). Importantly, vegetables are the most promising CO_2 mitigatory option in the presence of biochar amendment. In contrast, biochar application at all crop duration significantly promoted the CO_2 emission from soil. For N_2O, maize was followed by rice and fallow exhibited significant mitigation potential under following biochar amendment. However, crop duration did not show any significant effect since overall effect size was near to zero (*RR* = 0.067, 95% CI = -0.427, 0.56) (Figure 6 (IIb)). According to the CH_4 meta-analysis, biochar application, either at crop type or crop duration scale, significantly promoted CH_4 mitigation with the exception of fallow, marginally giving higher CH_4 emission trend (Figure 6 (III a, b).

3.7.Biochar application and GWP

Biochar amendment significantly increased global warming potential (GWP) (*RR* =0.24, 95% CI = -0.005, 0.536) in research studies that measured the emission fluxes of the three most potent GHGs (Figure 7). A closer look at the variables, including biochar feedstock, application rate, experimental conditions, pyrolysis temperature, and crop type showed significant impact of biochar application on the GWP. In terms of feedstock, lignocellulosic waste had the highest GWP (*RR* = 0.512, 95% CI = 0.359, 0.665) compared to other feedstocks (Figure 7 (a)). Importantly, biochar application rate (⩽ 30 T ha^{-1}) significantly decreased GWP (*RR* = -0.237, 95% CI = -0.46, -0.012) (Figure 7 (b)). Nevertheless, incubations studies showed significantly greater GWP to biochar addition responses (*RR* = 0.395, 95% CI = 0.296, 0.494; however, field (*RR* = -0.037, 95% CI = -0.104, 0.031) and pot (*RR* = -0.055, 95% CI = -0.269, 0.159) studies showed reduction in GWP (Figure 7 (c)). On the other hand, maximum and minimum pyrolysis temperatures showed an accelerated and repressed effect on GWP, respectively (Figure 7 (d)). For crop type, according to our meta-analysis results, the fallow system was the major contributor of GWP, while rice and wheat cropping systems were accounted for the reduction in GWP after biochar application (Figure 7 (e)). However, with the realization that few research studies were reported fluxes of all three GHGs after biochar application, these results were likely affected by publication biases, and therefore should be interpreted cautiously.

4. Discussion

In line with plethora of previous literature, our meta-analysis showed that GHGs emissions from biochar amended agricultural soils vary widely, owing to differences in biochar physicochemical attributes (i.e. feedstock type, pyrolysis temperature, C: N ratio, pH); soil edaphic factors (i.e. C: N ratio, pH, texture); different experimental conditions (pot, incubation, field) as well as crop types and biochar application rate (Fidel et al., 2019; Levesque et al., 2020; Rittl et al., 2018; Shakoor et al., 2021c; Subedi et al., 2016; Tarin et al., 2021). In general, while feedstock type demonstrated significant mitigation for CH_4, it led to significantly higher CO_2 and N_2O emissions (Figure 1). Indeed, there is an imperative evidence that biochar processing, including types of feedstock material and pyrolysis temperature conditions, can have significant impact on surficial attributes, physical and molecular composition of biochar, and thus affect soil CO_2, N_2O, and CH_4 emissions (Das et al., 2021; Deng et al., 2021; Nzediegwu et al., 2021).

Apparently, stronger emissions response of CO_2 and N_2O following biochar application is related to soil organic content, implying that biochar mediated increased substrate availability drives higher microbial activities associated with CO_2 and N_2O production (Sheng et al., 2010; Liu et al., 2016). Furthermore, we found that lignocellulosic biochar contributed to higher CO_2 emissions, which could be attributed to increased C mineralization of native organic matter and/or abiotic release of biochar bound C (Fidel et al., 2017; Jones et al., 2011; Li et al., 2018).

Soil amended with manure derived biochar was a greater CO_2 sink into the soil, which may be due to the pyrolysis linked higher stability of manure biochar that can reduced C mineralization from soil (Gascó et al., 2016). Furthermore, biochar pyrolysis temperature is another factor that can strongly affect both magnitude and direction of C mineralization in soil (Chen et al., 2021). Pyrolysis at low temperature ($\leqslant$ 600 ^{o}C) could increase the microbial responses to decompose the organic material, resulting in a higher rate of CO_2 release from agricultural soils (Chan et al., 2008; Hale et al., 2012). In contrast, high pyrolysis temperature biochar may accumulate several toxic functional groups, which therefore suppressed the soil microbial activity leading to lower rates of CO_2 emissions (Nakajima et al., 2007; Figure 1 (Ib)).

Based on our meta-analysis N_2O estimates, manure derived biochar is the major contributor of cropland N_2O emissions, whereas biosolids derived biochar demonstrates significant N_2O mitigation (Figure 1 (IIa)). Consistent with some recent results reported by Grutzmacher et al., (2018) and Thers et al., 2020), manure based biochar is known to be a richer N source than lignocellulosic biochar, acting as a major N_2O emission source from croplands. In agricultural soils, both denitrification and nitrification are the rate limiting steps of N_2O emissions (Nelissen et al., 2014; Lévesque et al., 2018), biochar related suppression in N_2O emissions as evident in our meta-analysis could be driven by a regulated decrease in the denitrification activity often mediated by biochar amendment (Case et al., 2015; Liu et al., 2020). Overall, the greater CH_4 mitigation response from biochar feedstock (Figure 1 (IIIa)) suggests that different feedstock sources may exhibit varying methanogen and methanotrophic activities due to differential affinity for CH_4 metabolism and consequently, resulting in lower emissions (He et al., 2017; Nguyen et al., 2020). Another significant finding related to CH_4 mitigation, which was also found in the biochar-methane meta-analysis by Jeffery et al (2016), was the marginally lower CH_4 emissions from cropland soils amended with biochar produced through high temperature pyrolysis (Figure 1 (IIIb)). This is perhaps to be expected, as high temperature feedstock combustion produces biochar with few labile constituents, which did not provide much to the microbes as a substrate upon application (Bruun et al., 2011; Song et al., 2016).

The studies in our meta-analysis on biochar amendment to different experimental conditions showed significant variation in the cropland CO_2 emissions (Figure 2 (I)); with incubated soil demonstrating a higher CO_2 release compared to field and pot experiments (He et al.,2017). Conceivably, from a CO_2 emission perspective in short duration experiments (i.e., soil incubation), bio-accessible C fraction together with an increased surface area of biochar particle could provide niche support for microbes (Chia et al., 2014; Pokharel et al., 2020), thereby accelerating rate of C mineralization and eventually higher soil CO_2 flux (Zimmerman et al., 2011). With regard to N_2O emissions, we discovered that biochar amendment under field conditions resulted in lower emissions than pot and incubation studies (Figure 2 (IIa)). While Song et al. (2016) also reinforced these findings, when field application of biochar had shown a considerable (up to 16%) reduction in N_2O emissions from cropland soil. The soluble base cations in biochar are most likely a critical component because they neutralize soil acidity by increasing pH, and consequently changing the product stoichiometry of the denitrification process favoring maximum conversion of N_2 (dinitrogen) to N_2O emissions (Cayuela et al., 2013; Van Zwieten et al., 2010; Yuan et al., 2011). Biochar application also displayed strong mitigation effects on CH_4 emissions in field experiment (Figure 2 (IIIa)). This contradicts the meta-analysis estimates of Liu et al (2016), who found that biochar application in cropland soils, particularly under field conditions, promotes CH_4 emissions. Low biochar application rates and high labile C lixiviation due to high precipitation could be involved in the reduction of CH_4 emissions under field conditions (Spokas and Reicosky, 2009).

Soil texture is another important factor that is expected to have an influence on widespread biochar efficiency. In our meta-analysis, biochar application to fine texture soils significantly increased CO_2 emissions, coarse soils had no significant effects on N_2O but exhibited stronger mitigation for CH_4 (Figure 2 (Ib, IIb, IIIb). Similar to our analysis, Cayuela et al., (2014) also reported no effect of biochar on N_2O emissions in coarse texture soil, although these effects were best predicted by soil moisture conditions. Aside from the conditioning effect of biochar, which improves soil aeration and CH_4 oxidation, shifts favoring higher methanogens to methanotrophs ratios may be the primary processes causing the reduction in CH_4 emissions from coarse soils (Feng et al., 2012; He et al., 2017).

Biochar application rates are the most influential variable in determining the extent of GHGs emissions as well as its probable effects on crop yield. In our meta-analysis, lower rates of biochar had strong effects on CO_2 emissions, while N_2O emissions were significantly affected by a higher biochar rate, whereas 20 t ha^{-1} and 40 t ha^{-1} showed a strong effect on CH_4 emissions. For crop yield, effects were consistent across different biochar rates (Figure 3). According to our results, 30 t ha^{-1} was the optimum biochar application rate where significant reduction and good crop yield were observed.

The physicochemical properties (e.g., pH and C: N) of biochar modify nutrient and C dynamics in the soil, and thereby regulate e physical protection of soil microorganisms against predators (Jindo et al., 2014). In this meta-analysis, CO_2 emissions response showed significant increase at all biochar pH (Figure 4 (Ia)).

Previously, Liu et al. (2016) found that CO_2 emissions can vary with biochar pH and reported the highest CO_2 emissions with < 7 biochar pH. These findings were also consistent to Crombie et al. (2015), who showed that biochar with pH < 7 increased C mineralization and consequently CO_2 emissions. On the other hand, acidic biochar showed strong N_2O mitigation potential, whereas acidic biochar increased CH_4 emissions from agricultural soils (Figure 4 (IIa, IIIa). Biochar pH is a key regulator of several soil biochemical processes, including nitrification and denitrification. With acidic biochar, pH-dependent changes in the denitrification and/or nitrification processes may be a key factor governing differences of soil N_2O emissions (Li et al., 2018; Aamer et al., 2020). Furthermore, biochar application effects on CH_4 emissions reduced with biochar pH, probably suggesting a dynamics shift in the activity of relevant soil microbial communities, from methanogenic to methanotrophic bacteria (Anders et al., 2013).

Biochar C: N ratio also an important parameter that significantly influences the GHGs emissions. Our study showed a significant reduction effect on CO_2 emissions with $\leqslant$ 50 C: N ratios to a noticeable positive response to N_2O emissions with $\leqslant$ 150 C: N ratios, however biochar with lower ($\leqslant$ 50) and higher (> 300) C: N ratios had strong mitigation effects on CH_4 emissions (Figure 5 (Ia, IIa, IIIa)). In another study, the biochar C: N ratio had shown strong positive correlation with CO_2 emissions (Sistani et al., 2019). Biochar amendment with higher C: N ratios significantly decreased N_2O emissions due to microbial N immobilization (Cayuela et al., 2010). Consequently, very less amount of soil N is available for microbial processes for N_2O emissions. Also, Feng et al. (2012) discovered strong CH_4 mitigation when cropland soils were amended with biochar at lower C: N ratio.

Soil pH and C: N ratios can also have significant influence on GHGs emissions budget of biochar amended cropland soils. In this meta-analysis, responses of CO_2 emissions to biochar amendment showed the highest emissions in neutral soil pH, although it significantly reduced soil N_2O and CH_4 emissions (Figure 4 (Ib, IIb, IIIb). For N_2O and CO_2, emission rates elevated following biochar applications to given soil C: N ratios; however, CH_4 showed a decreasing pattern of emissions (Figure 5 (Ib, IIb, IIIb). Consistent with these observations, Yang et al. (2017) found that proportional decreases in CH_4 emissions following biochar application are primarily driven by pH dependent methanotroph activity. By altering soil conditions (e.g., pH, C: N ratio), literature to date implies that biochar can mitigate CH_4 (Qin et al., 2016), although the effect size is often site and biochar type specific (Chen et al., 2017). Others have noted with high soil C: N ratio (e.g., > 10), biochar application could result in higher CO_2 and N_2O emissions from cropland soils (Lin et al., 2015; Zhang et al., 2016; Mechler et al., 2018). It appears that biochar modifies soil C: N ratio, which stimulates soil microbial activity and leads to higher GHG emissions (Muñoz et al., 2019). Soils with low C: N ratio offer significant CH_4 mitigation potential after being amended with biochar (Novais et al., 2018).

The crop species and study duration also played an important role in the differences found in GHGs emissions (Huang et al., 2018). In our meta-analysis, crop type and study duration showed significant effect on GHGs emissions (Figure 6). Our results suggest, fallow soils significantly increased CO_2 emissions after biochar application. Biochar amendment to bare soil could inhibit the breakdown of soil organic C content, which could prompt lower CO_2 emissions (Jones et al., 2011). Similarly, Zhang et al. (2012) also found that biochar application significantly decreased GHGs emissions from agricultural soils In this meta-analysis, crops with > 725 days had displayed strong mitigation effect on CO_2, N_2O, and CH_4 emissions (Figure 6 (Ib, IIb, IIIb)). Long term effects of biochar may decrease and/or resist microbial activities (Hardy et al., 2019). Therefore, long experimental studies, receiving biochar as a fertilizer source, can mitigate the GHGs emissions.

Global warming potential (GWP) is a basic index to foresee the probability of GHGs related impact on the lifetime and radiative power of these species (IPCC, 2013). Overall, the application of biochar significantly enhanced the GWP of GHGs emissions (Figure 7) and these findings are in accordance with Yang et al., (2017), Tan et al., (2019), and Shen et al., (2014). In recent times, agricultural land uses have shown its explicit effects on soil C and N dynamics across ecosystems; more importantly, these land use changes trade off climatic vulnerabilities revealed by emission fluxes of the three most potent GHGs i.e., CO_2, N_2O, and CH_4. Agricultural soils released a significant amount of GHGs into the atmosphere, accounting for nearly one-fifth of the annual increase in the total radiative power (Cole et al., 1997; He et al., 2017; Shakoor et al., 2021a). According to the FAO (2014), GHG emissions from croplands were approximately 5.3 Pg at the start of the last decade. While agriculture intensification and associated management practices have riled the ecological stability with rising GWP (Shang et al., 2011).

5. Limitations, suggestions and concluding remarks

In this meta-analysis, most of the experiments had been studied in China, Europe, and North America (Figure S1). There remains a lack of experimental studies in other continents, like South America, South-East Asia, Africa and Australia. Therefore, long-term experimental research studies are needed with biochar application in these regions. Most of the research studies that are included in this study reported no changes in physicochemical properties of soil (C: N and pH) after biochar amendment. Therefore, research on these aspects of soil should be prioritized to better understand the relationship between the application of biochar and GHGs emissions.

Following on, research studies on biochar application and GHGs emissions should systemically include complete biochar physicochemical properties (i.e. pH, C: N ratio, bulk density, BET surface area, EC, particle size, potential toxicity and adsorption capacity), origin and chemical properties of feedstock, pyrolysis conditions (i.e. temperature, rector type and exposure time) and physicochemical properties of soil such as pH, bulk density, texture, EC, total N, total organic C, available phosphorus and potassium contents.

This meta-analysis provided a comprehensive and quantitative synthesis of biochar application, biochar chemical properties and soil attributes effects on GHGs emissions. According to the evidence presented in this meta-analysis, switching from conventional to biochar based agriculture systems has the potential to reduce gaseous emissions from cropland soils. Across range of biochar feedstock, we found that lignocellulosic derived biochar is the major contributor of CO_2 emissions. Similarly, manure derived biochar significantly increased the N_2O emission while showing strong CO_2 mitigation potential. In case of CH_4, wood derived biochar showed the highest mitigation potential from cropland soils. High pyrolysis temperature ($> 600\ ^{\circ}C$) was another key performance indicator of biochar, which demonstrated a strong mitigation effect on CO_2 and CH_4 emissions, though the reduction in N_2O emissions was predominantly from biochar produced at low pyrolysis temperature ($\leqslant 500\ ^{\circ}C$). Biochar application significantly reduced GHGs emissions (i.e., CO_2, CH_4) from field experiments, implying stronger merits for its application in cropland soils. However, the lack of biochar application related mitigation effects on N_2O emissions is quite disconcerting, emphasizing the necessity of biochar-N_2O focused research in future. Furthermore, biochar use in fine textured soils resulted in significant CO_2 emissions; however, coarse soils significantly reduced N_2O and CH_4 emissions.

Based on application rate estimates, biochar amendment at $\leqslant 30$ (t ha^{-1}) was the most effective application rate for mitigating GHGs emissions while maintaining good crop yield potential. Certainly, the effects of biochar amendment on CO_2, N_2O, and CH_4 emissions were considerably dependent on physiochemical properties of biochar (pH and C: N ratio) and soil (pH, textural class, and C: N ratio), indicating that these important factors must be fully considered before applying biochar to cropland soil for the mitigation of GHGs emissions.

Hence, this meta-analysis suggests that experimental strategies, such as choice of the biochar feedstock, pyrolysis temperature, and appropriate application rate, should be carefully planned in order to mitigate GHGs emissions from agricultural soil.

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Declaration of competing interest

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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Figure

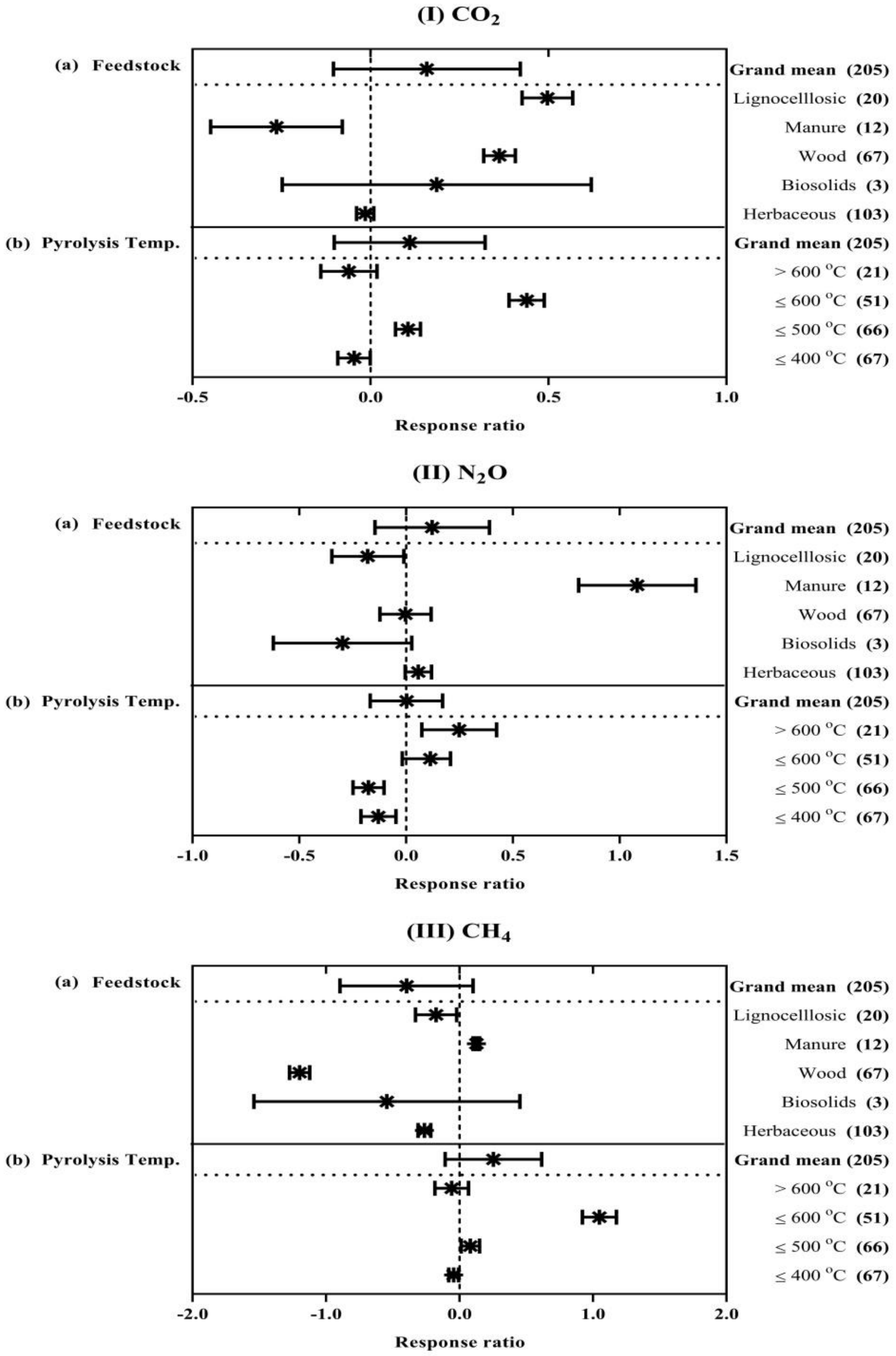

(I) CO_2
(a) Feedstock
Grand mean (205)
Lignocelllosic (20)
Manure (12)
Wood (67)
Biosolids (3)
Herbaceous (103)
(b) Pyrolysis Temp.
Grand mean (205)
> 600 °C (21)
≤ 600 °C (51)
≤ 500 °C (66)
≤ 400 °C (67)
-0.5
0.0
0.5
1.0
Response ratio
(II) N_2O
(a) Feedstock
Grand mean (205)
Lignocelllosic (20)
Manure (12)
Wood (67)
Biosolids (3)
Herbaceous (103)
(b) Pyrolysis Temp.
Grand mean (205)
> 600 °C (21)
≤ 600 °C (51)
≤ 500 °C (66)
≤ 400 °C (67)
-1.0
-0.5
0.0
0.5
1.0
1.5
Response ratio
(III) CH_4
(a) Feedstock
Grand mean (205)
Lignocelllosic (20)
Manure (12)
Wood (67)
Biosolids (3)
Herbaceous (103)
(b) Pyrolysis Temp.
Grand mean (205)
> 600 °C (21)
≤ 600 °C (51)
≤ 500 °C (66)
≤ 400 °C (67)
-2.0
-1.0
0.0
1.0
2.0
Response ratio

Figure. 1 Effect of (a) biochar feedstock type and (b) pyrolysis temperature (°C) on I) CO_2, II) N_2O and III) CH_4 emissions from agricultural soils. Parentheses numbers indicate the number of observations and error bars represent with 95% confidence intervals.

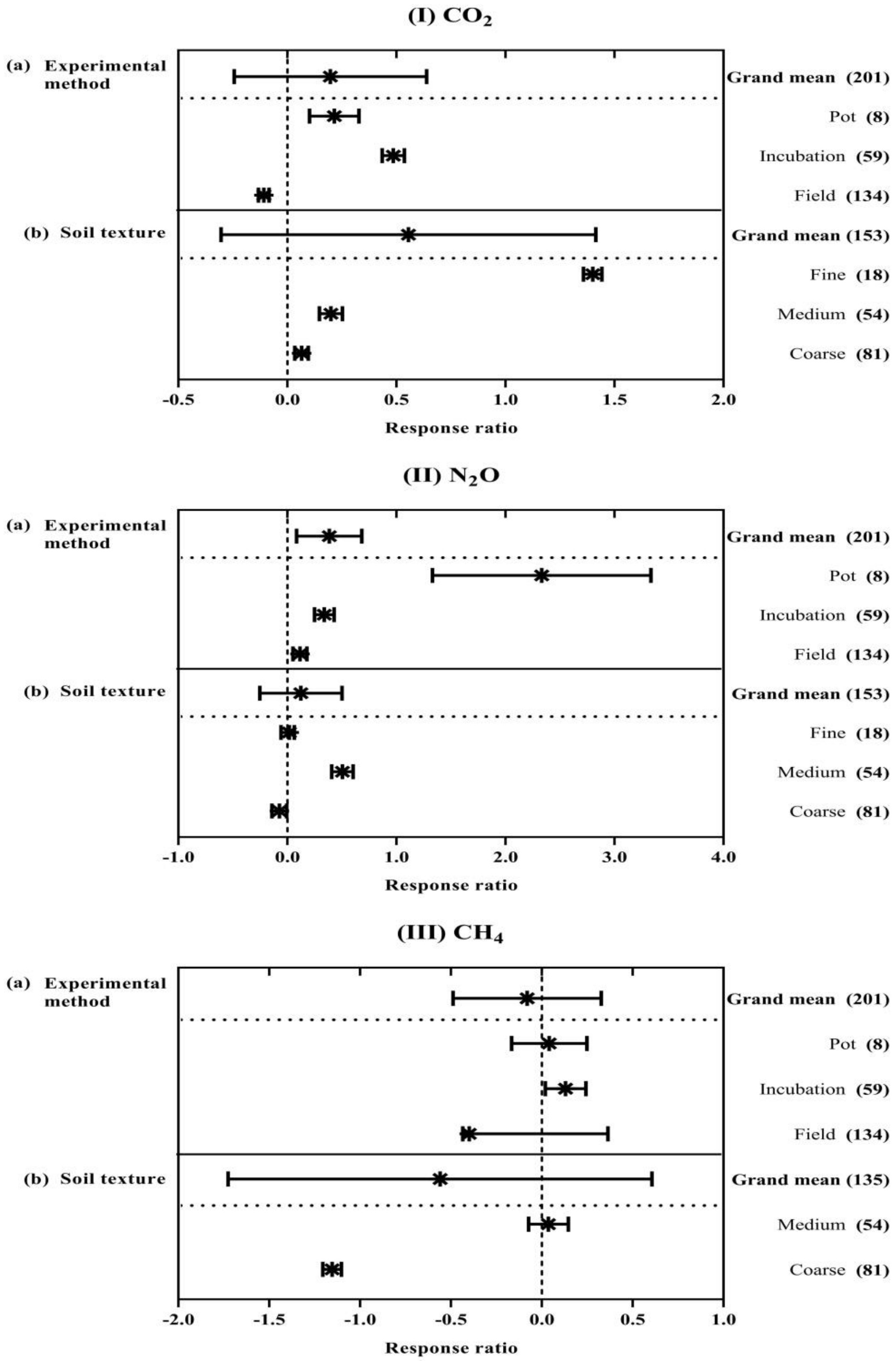


Figure. 2 Response ratios of I) CO_2, II) N_2O and III) CH_4 emissions to biochar application depended on (a) experimental method and (b) soil texture. Parentheses numbers indicate the number of observations and error bars represent with 95% confidence intervals.

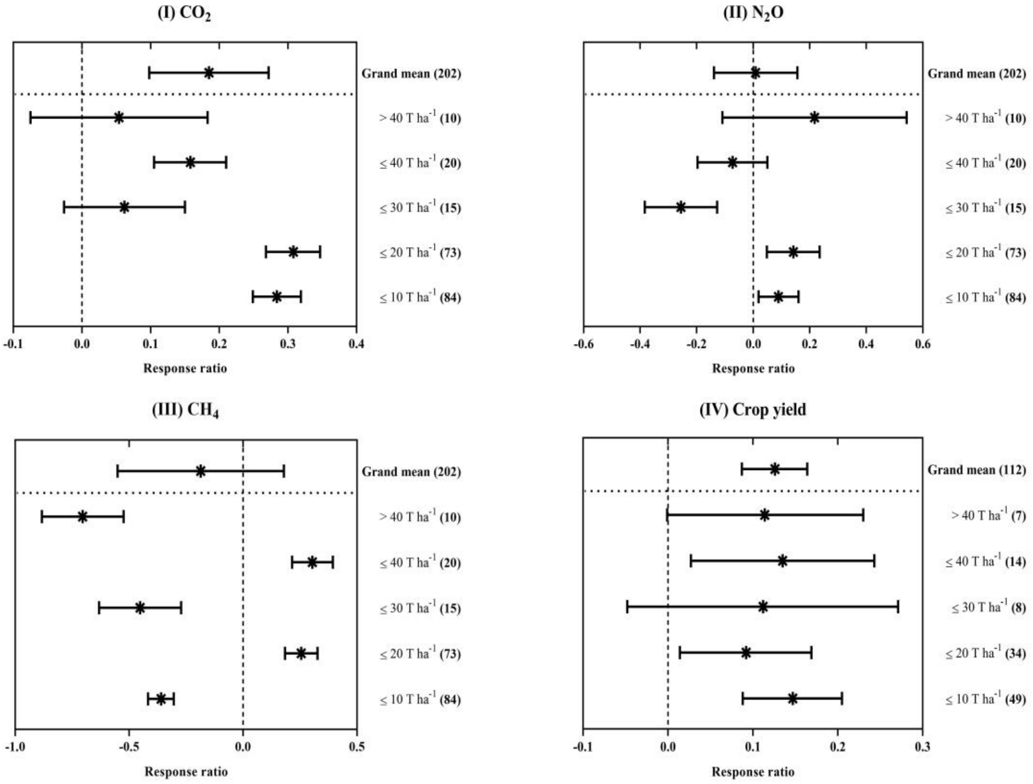


Figure. 3 Influence of biochar application rate (T ha^{-1}) on I) CO_2, II) N_2O, III) CH_4 emissions and IV) crop yield (T ha^{-1}). Parentheses numbers indicate the number of observations and error bars represent with 95% confidence intervals.

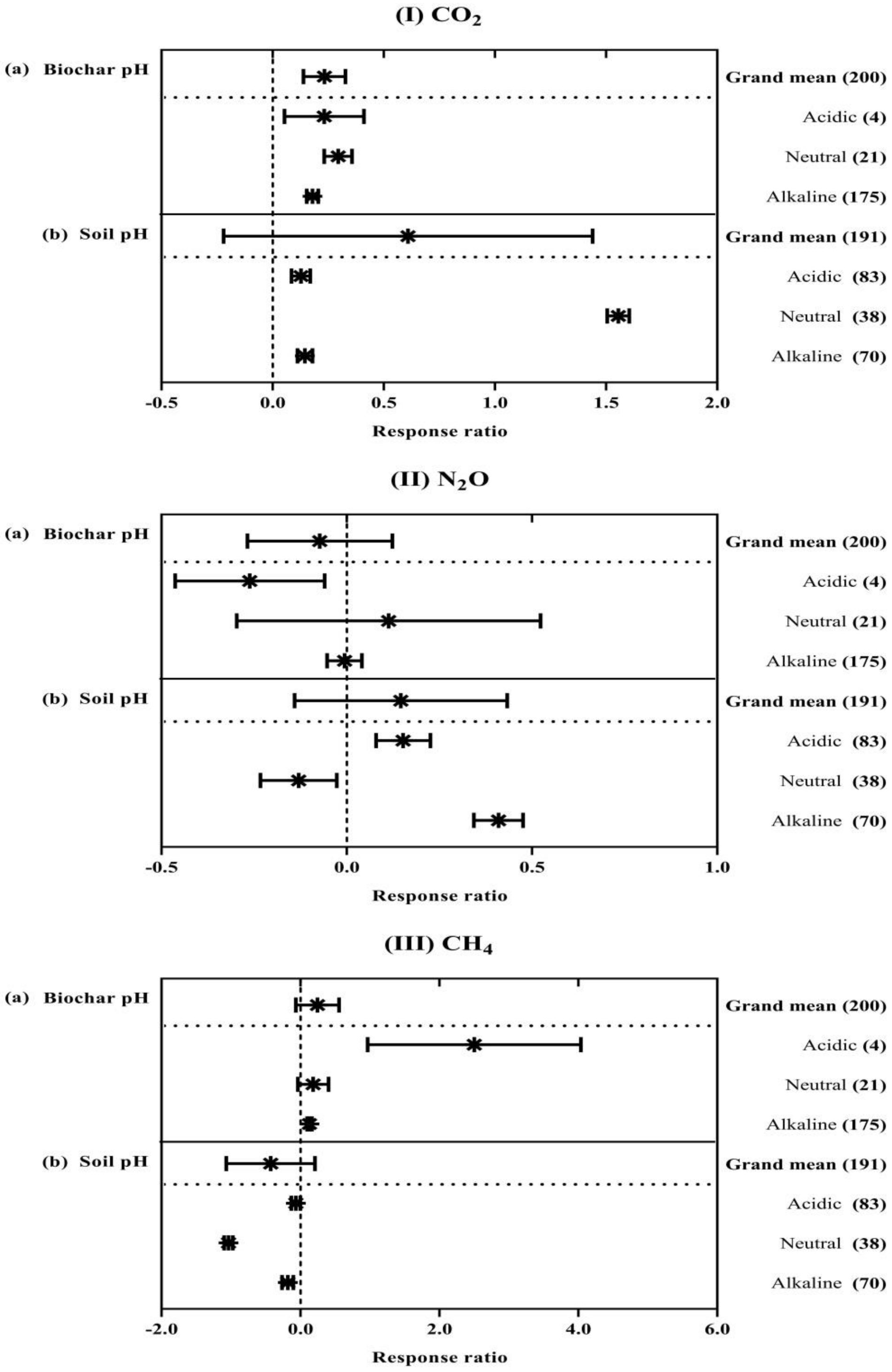


Figure. 4 Effect of biochar amendment on I) CO_2, II) N_2O and III) CH_4 emissions differed with (a) biochar pH and (b) soil pH. Parentheses numbers indicate the number of observations and error bars represent with 95% confidence intervals.

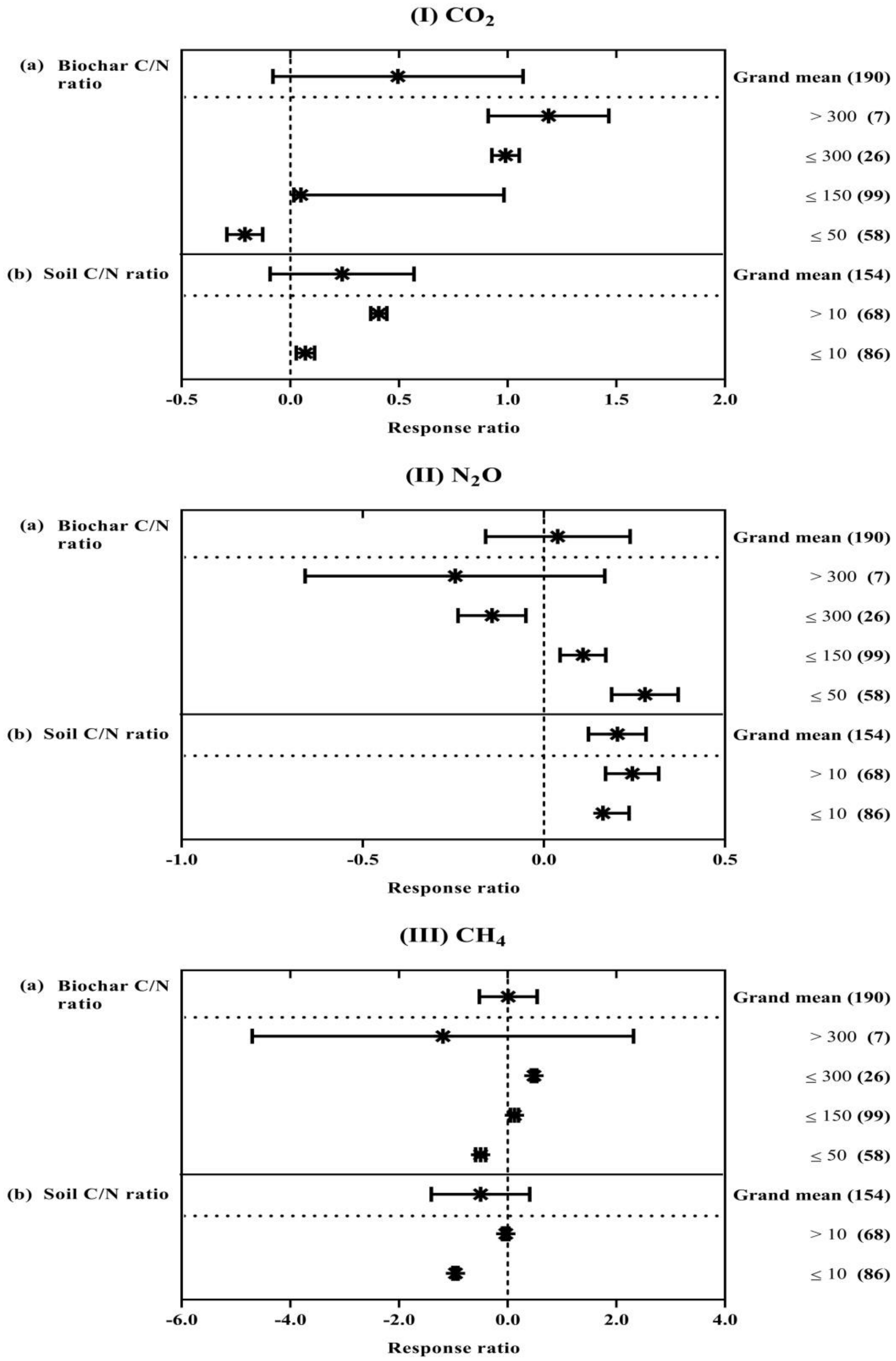


Figure. 5 Emissions of I) CO_2, II) N_2O and III) CH_4 after biochar application from agricultural soils affected by (a) biochar C: N ratio and (b) soil C: N ratio. Parentheses numbers indicate the number of observations and error bars represent with 95% confidence intervals.

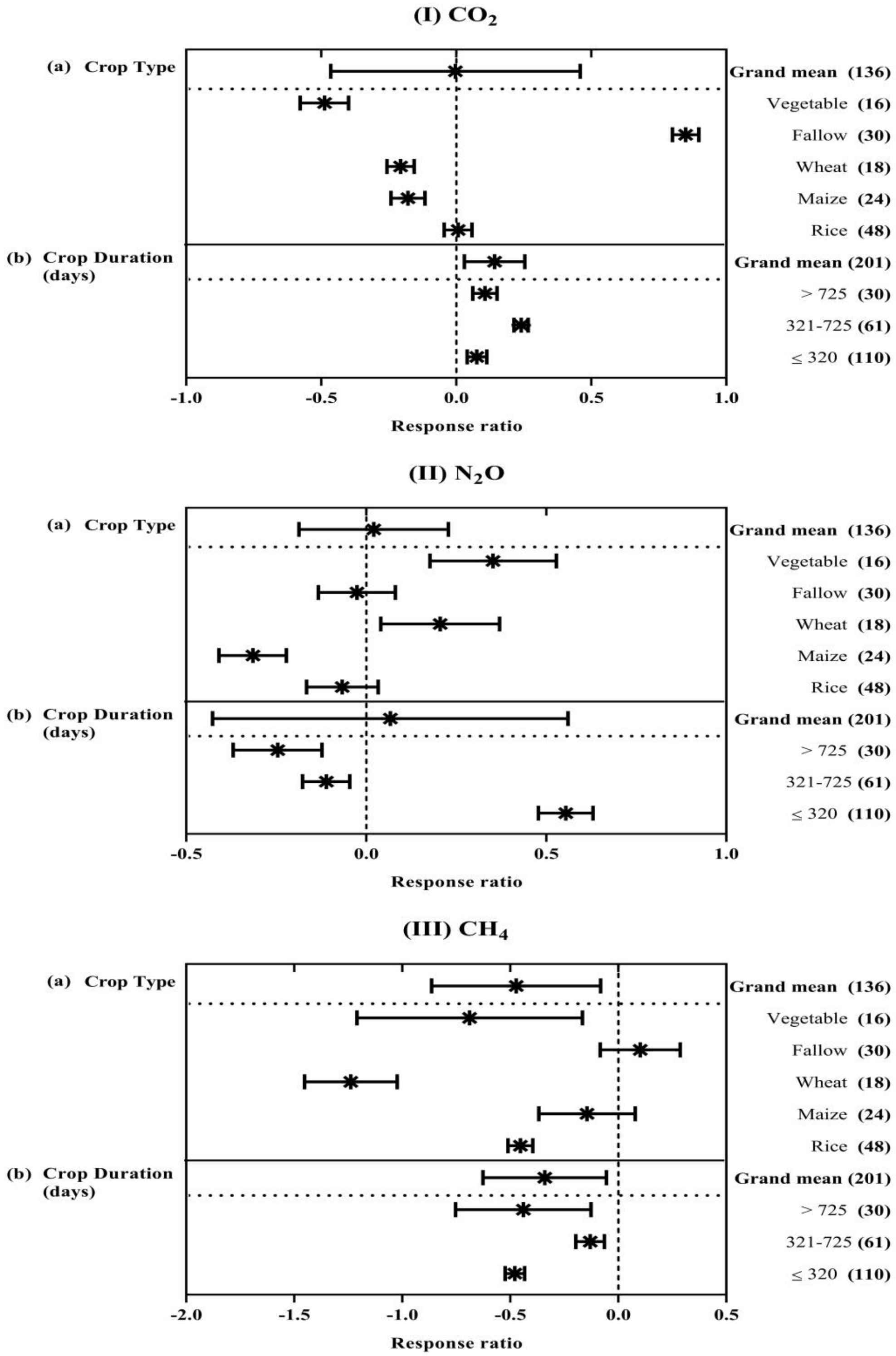


Figure. 6 Response ratios of I) CO_2, II) N_2O and III) CH_4 emissions to biochar application influenced by (a) crop type and (b) crop duration (days). Parentheses numbers indicate the number of observations and error bars represent with 95% confidence intervals.

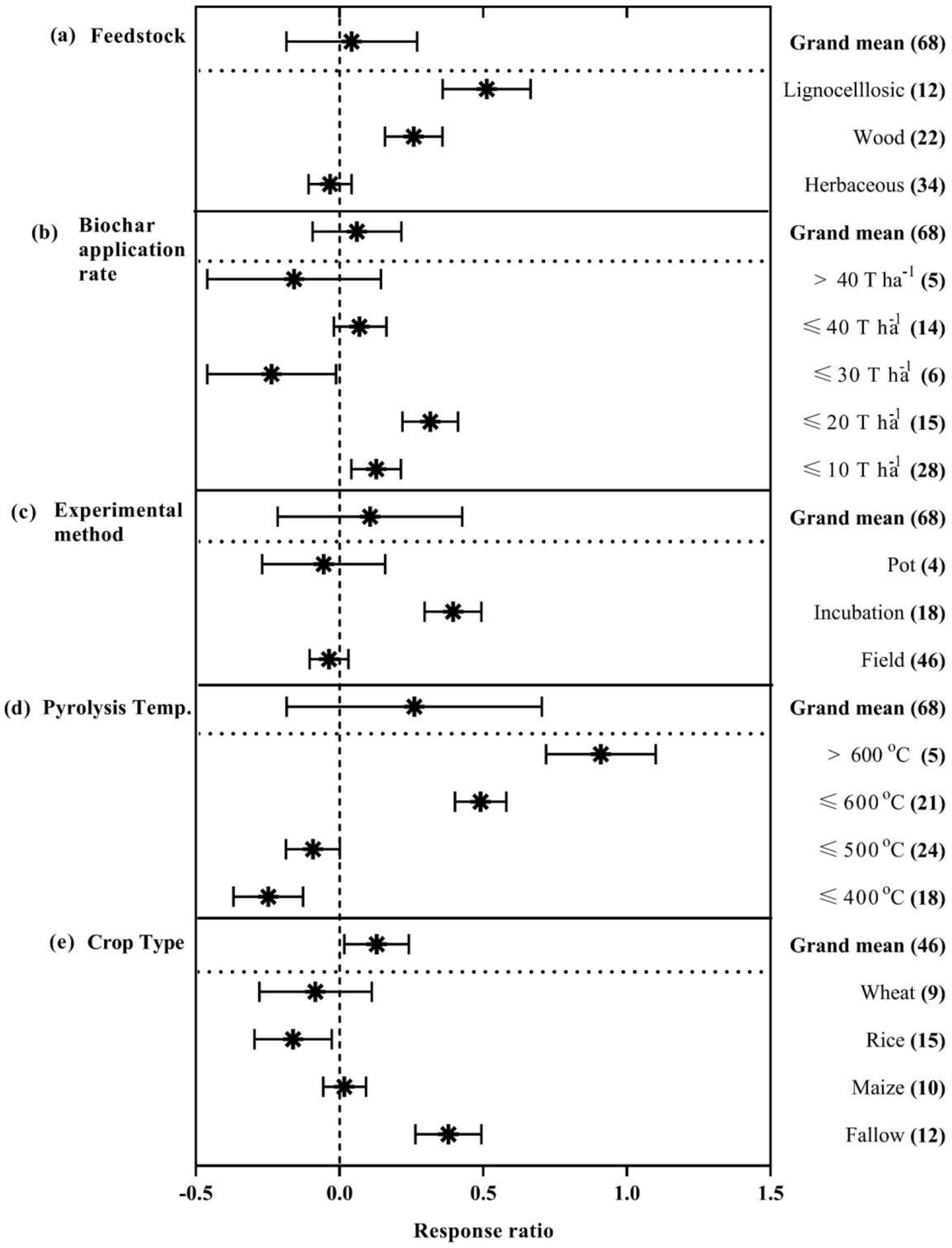
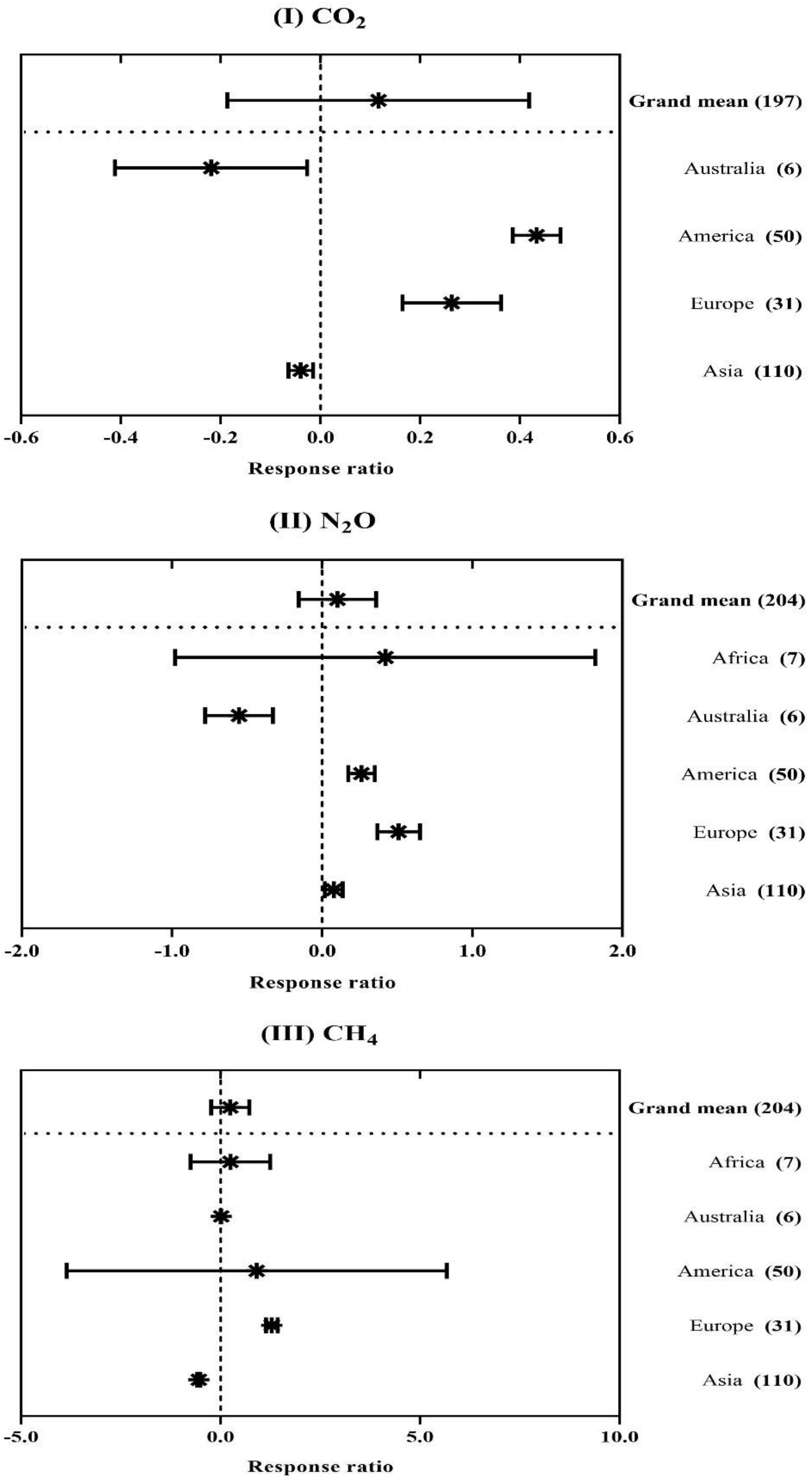


Figure. 7 Effect of biochar application on the global warming potential (GWP) of greenhouse gas (GHG) emissions under different parameters. Parentheses numbers indicate the number of observations and error bars represent with 95% confidence intervals.



Supplementary Figure S1 The effect of biochar application on I) CO_2, II) N_2O and III) CH_4 emissions by regions. Symbols represent mean effect sizes with 95% confidence intervals. The numbers in parentheses indicate the number of observations.

Table 1

Description of biochar feedstock, pyrolysis temperature, study duration, number of observation, experimental method, biochar application rate and physiochemical properties of soil and biochar included in this meta-analysis.

Study Reference	Region	Experimental method	Feedstock type	Pyrolysis temperature (ºC)	Biochar pH	Biochar C/N	Biochar application rate ($\mathrm{T\ ha^{-1}}$)	Soil pH	Soil C/N	Soil texture	Study duration (days)	Number of observations
Ali et al., (2013)	Asia	Field	Herbaceous	450	Alkaline	91.8	0-1	Acidic	-	Medium	≤320	4
Angst et al., (2014)	America[1]	Field	Biosolids	550	Alkaline	78.2	0-18.8	Neutral	-	Coarse	365	3
Zhang et al., (2012)	Asia	Field	Herbaceous	550	Alkaline	-	0-40	Acidic	-	-	385	8
Zhang et al., (2016)	Asia	Field	Herbaceous	450	Alkaline	79.1	0-40	Alkaline	10.75	-	≤320	12
Bayabil et al., (2016)	Africa	Field	Wood	-	-	-	0-12	-	-	-	-	6
Koga et al., (2017)	Asia	Field	Wood	800	-	103	0-40	Acidic	11.1	Medium	>725	4
Nguyen et al., (2016)	Australia	Field	Lignocellulosic	400	Alkaline	11.61	0-30	Acidic	12.7	Coarse Fine	≤320	6
Azeem et al., (2019)	Asia	Field	Herbaceous	350	Neutral	-	0-10	Alkaline	-	Coarse	600	12
Qin et al., (2016)	Asia	Field	Herbaceous	425	Alkaline	10.1	0-20	Neutral	9.68	Coarse	>725	32
Van Zwieten et al., (2013)	Australia	Field	Manure	550	Alkaline	13.64	0-10	Acidic	12	-	≤320	2
He et al., (2018)	Asia	Field	Herbaceous	450	Alkaline	42.5	0-15	Acidic	11.1	-	≤320	3
Puga et al., (2020)	America[2]	Field	Wood	400	Alkaline	78.8	0-20	Acidic	-	Fine	≤320	4
Thers et al., (2020)	Europe	Field	Herbaceous	750	Alkaline	92	0-15	Acidic	12.08	Coarse	402	5

Study Reference	Region	Experimental method	Feedstock type	Pyrolysis temperature (ºC)	Biochar pH	Biochar C/N	Biochar application rate (T ha^{-1})	Soil pH	Soil C/N	Soil texture	Study duration (days)	Number of observations
Basalirwa et al., (2020)	Asia	Field	Lignocellulosic	450	Alkaline	67.4	0-40	Acidic	11	Medium	≤320	8
Zhang et al., (2010)	Asia	Field	Herbaceous	400	Alkaline	79.1	0-40	Acidic	13.3	-	≤320	3
Karhu et al., (2011)	Europe	Field	Wood	400	-	101	0-9	Neutral	22.6	Medium	≤320	2
Ramlow et al., (2019)	America[1]	Field	Wood	550	Alkaline	478	0-25	Alkaline	12.4	Medium	≤320	2
Qi et al., (2020)	Asia	Field	Herbaceous	500	Alkaline	-	0-10	Acidic	-	-	≤320	4
Lévesque et al., (2018)	America[2]	Incubation	Wood	400-700	Alkaline	54.04-76.09	0-40	-	-	-	≤320	6
Scheer et al., (2011)	Australia	Field	Manure	550	Alkaline	44	0-10	Acidic	9.78	Medium	≤320	2
Shen et al., (2014)	Asia	Field	Herbaceous	400	Alkaline	38.3	0-48	Acidic	9-9.29	Coarse	≤320	6
Case et al., (2012)	Europe	Incubation	Wood	400	Alkaline	85.8	0-22	Alkaline	5.4	Coarse	≤320	9
Li et al., (2015)	Asia	Field	Herbaceous	400	Alkaline	83.3	0-40	Acidic	7.57	Fine	419	12
Liu et al., (2014)	Asia	Field	Herbaceous	500	Alkaline	72.12	0-48	Acidic	9	Coarse	370	3
Zhang et al., (2012)	Asia	Field	Herbaceous	452	Alkaline	79.1	0-40	Alkaline	10.5	-	≤320	3
Sistani et al., (2019)	America[1]	Field	Wood	600	Acidic	273.16	0-21.28	Acidic	9.3	Medium	331	4
Sriphirom et al., (2020)	Asia	Field	Wood	350	Alkaline	274.5	0-10	Neutral	9.55	Coarse	≤320	4
Sriphirom et al., (2020)	Asia	Field	Wood	350	Alkaline	274.5	0-10	Neutral	9.55	Coarse	≤320	4
Liu et al., (2019)	Asia	Field	Herbaceous	500	Alkaline	50	0-11.25	Alkaline	8.18	Coarse	>725	4

Study Reference	Region	Experimental method	Feedstock type	Pyrolysis temperature (ºC)	Biochar pH	Biochar C/N	Biochar application rate (T ha^{-1})	Soil pH	Soil C/N	Soil texture	Study duration (days)	Number of observations
Lin et al., (2015)	Asia	Field	Herbaceous	400	Alkaline	51.1	0-32	Alkaline	16	Coarse	370	8
Nelissen et al., (2014)	Europe	Incubation	Wood-Herbaceous	450-650	Neutral-Alkaline	85-617	0-20	Acidic	9.91	Medium	≤320	7
Aguilar-Chávez et al., (2012)	America[2]	Field	Biosolids-Wood	300	Alkaline	15	0-4.5	Alkaline	12	Corarse	≤320	4
Subedi et al., (2016)	Europe	Incubation	Manure-Wood	400-1000	Alkaline	9-335.4	0-20	Alkaline-Acidic	9.1-8	Medium	≤320	12
Abbruzzini et al., (2019)	America[2]	Pot	Herbaceous	450	Alkaline	56.2	0-19	Acidic	9.1	Medium	≤320	4
Sun et al., (2020)	Asia	Field	Herbaceous	450	Alkaline	83.3	0-40	Alkaline	12.7	Medium	≤320	3
Wang et al., (2012)	Asia	Pot	Lignocellulosic	450	Alkaline	75	0-50	Alkaline	11.4	Medium	≤320	6
Verhoeve et al., (2014)	America[1]	Field	Lignocellulosic-Wood	550-900	Alkaline	7.9-9.7	0-10	Neutral	-	Coarse	725	6
Singla et al., (2014)	Asia	Pot	Manure	330	Alkaline	9.3	0-7.9	Acidic	10.7	-	≤320	2
Tan et al., (2019)	Asia	Field	Lignocellulosic	450	Alkaline	79.8-112.7	0-50	Alkaline	20	-	≤320	5
Watanabe et al., (2014)	Asia	Field	Herbaceous	800	-	98.8	0-20	-	4.5-9	-	≤320	12
Wells et al., (2014)	Europe	Incubation	Wood	980	Alkaline	200	0-6	-	-	Coarse	≤320	2
Fungo et al., (2014)	Africa	Fallow	Herbaceous-Wood	350-550	Alkaline	86-437.8	0-15	Acidic	-	-	≤320	5
Schimmelpfennigt al., (2014)	Europe	Incubation	Herbaceous	600	Acidic-Neutral-Alkaline	152-399.5	0-16	Acidic	10.6	-	≤320	4

Study Reference	**Region**	**Experimental method**	**Feedstock type**	**Pyrolysis temperature (ºC)**	**Biochar pH**	**Biochar C/N**	**Biochar application rate (T ha^{-1})**	**Soil pH**	**Soil C/N**	**Soil texture**	**Study duration (days)**	**Number of observations**
Thomazini et al., (2015)	America[1]	Incubation	Wood	550	Neutral	140	0-10	Acidic-Neutral-Alkaline	-	Medium	≤320	20
Gomez et al., (2014)	America[1]	Incubation	Wood	550	Alkaline	253	0-20	Alkaline		Coarse-Fine-Medium	365	20
Sun et al., (2014)	Asia	Incubation	Herbaceous	450	Alkaline	83.3	0-30	Acidic	8.28-33.4	Fine-Medium	365	4

[1] North America; [2] South America

Supplementary Table S1 Statistical results were reported as total heterogeneity (Q_t) with observations (N) in effect sizes among studies from continuous randomized-effects model meta-analysis for CO_2, N_2O and CH_4 emissions.

Study parameter	**N[a]**	**Q_t**	**N[b]**	**Q_t**	**N[c]**	**Q_t**
Feedstock type	205	350.313***	205	66.869***	205	1178.006***
Pyrolysis temperature	205	230.608***	205	37.339***	205	266.960***
Experimental method	201	436.387***	201	33.256***	201	91.709***
Soil texture	153	2608.904***	153	96.975***	135	373.049***
Biochar application rate (T ha^{-1})	202	50.345***	202	31.302***	202	311.515***
Biochar pH	200	11.490**	200	11.490	200	9.379*
Soil pH	191	2467.657***	191	78.602***	191	511.558***
Biochar C/N	190	817.176***	190	42.741***	190	350.824***
Soil C/N	154	139.686***	154	2.390	154	583.348***
Crop type	136	1296.540***	136	59.952***	136	72.592***
Crop duration	201	60.853***	201	206.817***	201	72.592***
Study region	197	323.197***	204	74.467***	204	1017.885***

[a]CO_2; [b]N_2O; [C]CH_4; Statistical significance of Q_t: *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$.