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RESEARCH ARTICLE

Increases in temperature response to CO_2 emissions in biochar-amended vegetable field soil

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Abstract

To explore the effects of biochar application on CO_2 and CH_4 emissions as well as the temperature response of CO_2 emissions, a 1-year experiment was conducted with three treatments (control; CF, chemical fertilizer only; BCF, biochar combined with chemical fertilizer) in a vegetable field. The results showed that (1) compared with CF, short-term application of biochar significantly enhanced the cumulative CO_2 emissions by 27.5% from a soil–plant system by increasing the soil microbial biomass (e.g., MBC) and C substrates (e.g., SOC); (2) lowest emissions of CH_4 were observed in the BCF treatment, and an increase in CH_4 consumption and reduced competition with NH_4^+ may be responsible for the significant reduction in CH_4 source strength in biochar-amended soil; and (3) activation energy (E_a) was identified as an important factor influencing the temperature sensitivity (Q_{10}) of CO_2 emissions. Fertilization (CF and BCF) reduced the average Q_{10} and E_a values of CO_2 emissions by 9.0–26.7% and 23.5–10.1%, respectively, relative to the control. In addition, the average E_a value in the BCF treatment (51.9 kJ mol^{-1}) was significantly higher than those in the control and CF treatments. The increase in Q_{10} and E_a values following biochar application possibly contributed to the supplementation of limited labile C and nutrients but highly resistant C following biochar application. Soil pH and crop cultivation may play key roles in influencing the change in E_a. Our study concludes that biochar amendment increased CO_2 emissions and temperature response of CO_2 emission from the soil–plant system while reducing CH_4 emissions.

Keywords Biochar · Greenhouse gas · Temperature sensitivity · Activation energy

Introduction

The changing climate was mainly induced by greenhouse gas (GHG) emissions, including carbon dioxide (CO_2) and methane (CH_4). In the last 20 years, CH_4 emission around the world increased by 10% (Jackson et al. 2020), and the emissions of GHG from agricultural ecosystems were about 5.24 Gt CO_2 equivalents $year^{-1}$, which contributed 11% of the total global anthropogenic emissions (Pearson et al. 2017). Hence, altering agricultural management schemes is warranted to reduce GHG emissions and mitigate climate change (Tang et al. 2021).

Turnover of soil organic carbon (SOC) was found as an important factor that largely influences the global climate change (Pan et al. 2004). For example, sequestration and mineralization of SOC are closely related to the GHG emissions (Lee et al. 2020). Fang et al. (2017) reported that global warming may lower the C sequestration potential. During the mineralization of SOC, temperature plays a vital role, which results in variability in the C pool (Criscuoli et al. 2019; Kan et al. 2020; Wang et al. 2019). The response to temperature changes, such as temperature sensitivity (Q_{10}, defined as the rate of change of soil CO_2 emission as a consequence of temperature increase of 10 ℃) (Kirschbaum

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Highlights
- Combination of biochar and N cannot offset the negative effect of biochar on soil CO_2 emissions.
- Short-term application of biochar showed a significant increase in CH_4 sink strength/reduction in CH_4 source strength.
- A lower value of E_a is responsible for the lower Q_{10} in soil treated with fertilizer.

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1995) and activation energy (E_a, defined as the necessary energy for reacting molecules to break and form new bonds after a collision) (Thiessen et al. 2013), could be used to evaluate the feedback intensity between CO_2 emission and global warming (Zhou et al. 2009), as well as the response of SOC to global warming (Fang et al. 2014). Generally, the value of Q_{10} increased with recalcitrance of decomposed substrates (Craine et al. 2010; Wang et al. 2019).

Biochar, as a soil amendment, has been incorporated into soil to improve soil properties and soil structure, increase nutrient availability, and microbial activities (Anderson et al. 2011; Criscuoli et al. 2014; Duan et al. 2020; Dai et al. 2021). As a stable amendment, biochar currently has been an attractive measure to enhance C sequestration on a long-term field scale (Singh et al. 2015; Kan et al. 2020). Therefore, there has been growing call to add biochar into soil to promote C sequestration and improve soil quality. However, in short periods of time (i.e., months), biochar will undergo structural changes, primarily the oxidation of surface, and can be utilized by microbes as a C source (Cheng et al. 2006; Zavalloni et al. 2011). As a result, biochar could be an ecosystem C source, instead of a sink, within a short-term period in soil. For example, Ameloot et al. (2013) determined that the increases in short-term CO_2 and N_2O emissions (117 days) were observed in biochar-amended soils due to the rapid degradation of labile compounds in the biochar (Zimmerman et al. 2011). Alternatively, new substrate addition would stimulate the "priming effects," defined as the changes in the mineralization of native soil organic matter (Kuzyakov et al. 2000; Kuzyakov 2010). The negative priming effects, such as reduced N_2O production and CH_4 oxidation, have been reported in soil treated with biochar (Spokas and Reicosky 2009; Wu et al. 2019; Duan et al. 2020) due to biochar's porous native and high affinity for natural organic matter (Kasozi et al. 2010; Zimmerman et al. 2011). However, biochar could also promote the mineralization of soil C due to the positive priming effect (Dong et al. 2018; Kan et al. 2020; Dai et al. 2021). Meanwhile, biochar incorporation can increase the root biomass, net photosynthesis, and grain yield, and then influence the net CO_2 emissions from the soil–plant system (Masto et al. 2013; Sun et al. 2017). Hence, the short-term response of greenhouse gas emissions to the biochar application in agricultural systems should receive more attention.

Exogenous C input (e.g., biochar) may alter the chemical recalcitrance of organic matter and environmental conditions, and result in a change in the temperature response of CO_2 emissions (Fang et al. 2014, 2017; Wang et al. 2019). According to the fundamental enzymatic kinetic theory, organic compounds with higher molecular weights showed lower rates of decomposition and higher values of Q_{10} and E_a relative to organic compounds with lower molecular weights. However, the decreases and increases in Q_{10} and E_a were observed in biochar-added soils (He et al. 2016; Fang et al. 2017; Pei et al. 2017; Wang et al. 2019). The contradictory results may be caused by the interactions of physical–chemical protection and substrate C quality change (Conant et al. 2011). Biochar application in a short-term period may introduce more C, including stable and labile C, which is related to the temperature response. However, most of previous studies on the temperature response to C emission were conducted in laboratory incubation, and more field works are necessarily needed.

Here, we hypothesized that biochar incorporated into soil would increase the gaseous C loss and temperature sensitivity of CO_2 emissions, especially in a short time period. In this study, we conducted a short-term vegetable cultivation experiment (approximately 1 year) to investigate the response of CO_2 and CH_4 emissions as well as the temperature sensitivity of CO_2 emissions to biochar amendment. The objectives of this study were (1) to explore the effects of biochar amendment on the soil CO_2 and CH_4 emissions, (2) to determine the temperature response of CO_2 emissions in biochar-amended soil, and (3) to try to identify key factors that influence C emissions and the temperature response of CO_2 emissions.

Materials and methods

Study site description

The experiment was conducted in the National Monitoring Station of Soil Fertility and Fertilizer Efficiency on Purple Soils (30°26′N, 106°26′E) in the Beibei District of Chongqing, southwestern China. The in situ soil is classified as Regosol in the Food and Agriculture Organization classification scheme (FAO 1988). The details of this trial site were described in the study of Huang et al. (2018, 2019). The basic property of soil is shown in Table 1.

Experimental design

Nine 2 m × 1 m plots were selected for this study from 2016 to 2017. Three treatments (one treatment per plot), including no fertilizer (control), chemical fertilizer only (CF), and biochar combined with chemical fertilizer (BCF), were arranged in a completely randomized design with three replicates (total 9 plots). The same amount of total nitrogen (N), phosphorus (P), and potassium (K) was applied in the CF and BCF treatments. Chemical fertilizers were applied as urea (N-eq, 46%), single superphosphate (P_2O_5-eq, 12%),

Table 1 Properties of background soil and biochar used in this study

	Soil	Biochar
pH	5.8	8.9
Organic carbon (g kg^{-1})	11.12 ± 0.66	625.8 ± 60.1
Total N (g kg^{-1})	0.82 ± 0.04	4.4 ± 0.6
Total P (g kg^{-1})	0.76 ± 0.05	0.97 ± 0.04
Total K (g kg^{-1})	20.7 ± 1.1	10.4 ± 0.3
C/N	13.6 ± 1.2	142.2 + 11.2
Available N (mg kg^{-1})	83.0 ± 3.6	
Available P (mg kg^{-1})	44.1 ± 4.1	
Available K (mg kg^{-1})	208.8 ± 23.6	
CEC (cmol(+) kg^{-1})	23.2 ± 1.5	
< 0.002 mm	31.30 ± 0.28%	
0.05–0.002 mm	39.10 ± 1.71%	
0.05–2 mm	29.60 ± 1.72%	
Soil texture	Clay loam	

Mean ± standard deviation ($n = 3$)

and muriate of potash (K_2O-eq, 60%). Biochar derived from rape straw was purchased from Sichuan Jiusheng Agricultural Technology Development Co. Ltd., China. The property of biochar is given in Table 1.

Four vegetable crops were grown in rotation during the experimental period from November 2016 to November 2017. The cultivated vegetable crops were lettuce (*Lactuca sativa* L. var. *angustana Irish*, November 2016 to January 2017), cabbage (*Brassica oleracea* L. var. capitata L., January 2017 to May 2017), chili (*Capsicum annuum* L., May 2017 to September 2017), and lettuce (*Lactuca sativa* L. var. *angustana* Irish, September 2017 to November 2017). In the CF treatment, the amount of chemical fertilizer was applied according to the Fertilization Guide for Major Crops in China (Zhang et al. 2009), as shown in our previous study (Huang et al. 2019). In the BCF treatment, 10 t hm^{-2} biochar was applied to soil before transplanting lettuce (October 20, 2016) and chili (May 5, 2017) for each addition according to our previous study (Huang et al. 2019). The deficient nutrients in the BCF treatment were supplemented with chemical fertilizer based on the same amount of total N, P, and K. Chemical fertilizers in the CF and BCF treatments were applied through basal fertilization and topdressing. The fertilization procedures were described in our previous study (Huang et al. 2019). The time schedule for fertilization and vegetable cultivation for different vegetables is described in Table S1.

Measurement of CO_2 and CH_4

The gases of CO_2 and CH_4 were sampled using the static closed chamber method during the experimental period. The setup of the chamber and the method of gas collection were given in the study of Huang et al. (2019). Briefly, gas samples were collected once every week (between 9:00 and 11:00) and every 2 or 3 days for 1 week following basal fertilizer and topdressing. After gas sample collection, the fluxes of CO_2 and CH_4 were measured simultaneously via the gas chromatography facility (Agilent 7890A; Agilent, Inc., USA). During the entire experiment, gas samples were collected 63 times in total. The calculations used to determine CO_2 and CH_4 fluxes and cumulative CO_2 and CH_4 emissions were similar to the study reported by Huang et al. (2019). Air and soil temperature (5 cm depth in soil) and the soil moisture content were recorded at the beginning and the end of sampling, and average of the two values was calculated. Because the greenhouse gas chamber measurements cannot exclude CO_2 emissions from plant roots, the CO_2 emissions in this study were the net CO_2 emissions from vegetable fields, which integrated soil respiration, belowground greenhouse gas emissions, and CO_2 assimilated by plants.

Soil sampling and measurements

Topsoil (0–20 cm) was sampled on November 23, 2017. In each plot, five soil cores were randomly sampled and mixed to form a pooled sample. The pooled samples were placed in the sterile plastic bags and transported to the laboratory. Meanwhile, soil bulk density was obtained via the cutting ring method. Sampled soil was thoroughly mixed and passed through a 2-mm sieve after all the visible roots and stones had been removed. Fresh soil was used for the analysis of soil dissolved organic carbon (DOC) and microbial biomass carbon (MBC), and the final concentrations of DOC and MBC were normalized by the dry mass of soil. The remaining soil was air-dried to measure the total SOC and soil pH.

Soil water–filled pore space (WFPS) was calculated according to the following equation (Li et al. 2013): WFPS = (gravimetric moisture × soil bulk density × 100) / [1 − (soil bulk density / 2.65)], with 2.65 g cm^{-3} of particle density.

Soil DOC content was extracted with a soil-to-water ratio of 1:10 (w/w), and the extracted solution was centrifuged and filtered through prewashed 0.45-μm cellulose acetate filters. All filtered solutions were measured via the Multi N/C® 2100 Analyzer (Analytik Jena, Germany) (Ghani et al. 2003). After being extracted by chloroform fumigation with 0.5 mol L^{-1} K_2SO_4, the extracts were used to measure the soil MBC content through the method of $K_2Cr_2O_7$ external heating with titrating $FeSO_4$ (Yang et al. 2008).

Temperature response

Temperature sensitivity (Q_{10}) and activation energy (E_a) of CO_2 emission were used to describe the relationship between temperature and CO_2 emission.

The Q_{10} was calculated with the following equation (Zhou et al. 2007; Chen et al. 2016):

$$y = a \cdot e^{bT} \tag{1}$$

$$Q_{10} = e^{10b} \tag{2}$$

where y is the flux of CO_2 over time (mg m^{-2} h^{-1}), and a and b are the exponential fit parameters. Parameter a is the intercept of CO_2 flux when the temperature is 0 ℃. T is the soil temperature (℃).

The activation energy was calculated using the exponential Arrhenius function according to Thiessen et al. (2013):

$$y = A \cdot e^{\frac{-E_a}{R \cdot T}} \tag{3}$$

where y is the flux of CO_2 over time (mg m^{-2} h^{-1}), A is the constant, E_a is the activation energy (J mol^{-1}), R is the universal gas constant (8.314 J mol^{-1} K^{-1}), and T is the soil temperature in Kelvin (K). In chemical kinetics, E_a is defined as the necessary energy for reacting molecules to break and form new bonds after a collision. To calculate the daily E_a, a maximum likelihood estimate of the slope of the linear regression of the natural logarithms of CO_2 flux against the reciprocal of absolute soil temperature was obtained. To estimate the average E_a during the experimental period, we multiplied the slope values by the gas constant R.

Statistical analysis

The data were statistically analyzed using SPSS 23.0 and Origin 8.5 software. The Kolmogorov–Smirnov test was used to test the normality of all data. Both parametric and nonparametric approaches were used to test the differences. For the normal distributed data, comparisons of data among treatments were performed by one-way analysis of variance analysis (ANOVA) in combination with the least significant difference (LSD) test. For non-normally distributed data, comparisons of data were performed by the Kruskal–Wallis test. After Bartlett's test of sphericity ($p < 0.05$), the variables related to soil properties, Q_{10}, E_a, and cumulative CO_2 and CH_4 emissions were subjected to principal component analysis (PCA) to identify key factors for Q_{10}, E_a, and cumulative CO_2 and CH_4 emissions using Origin 8.5. Automatic linear modeling was performed at the 95% confidence level using SPSS 18.0. Spearman's coefficient was used in the nonparametric correlation analysis. Statistical significance was determined at $p = 0.05$ and $p = 0.01$.

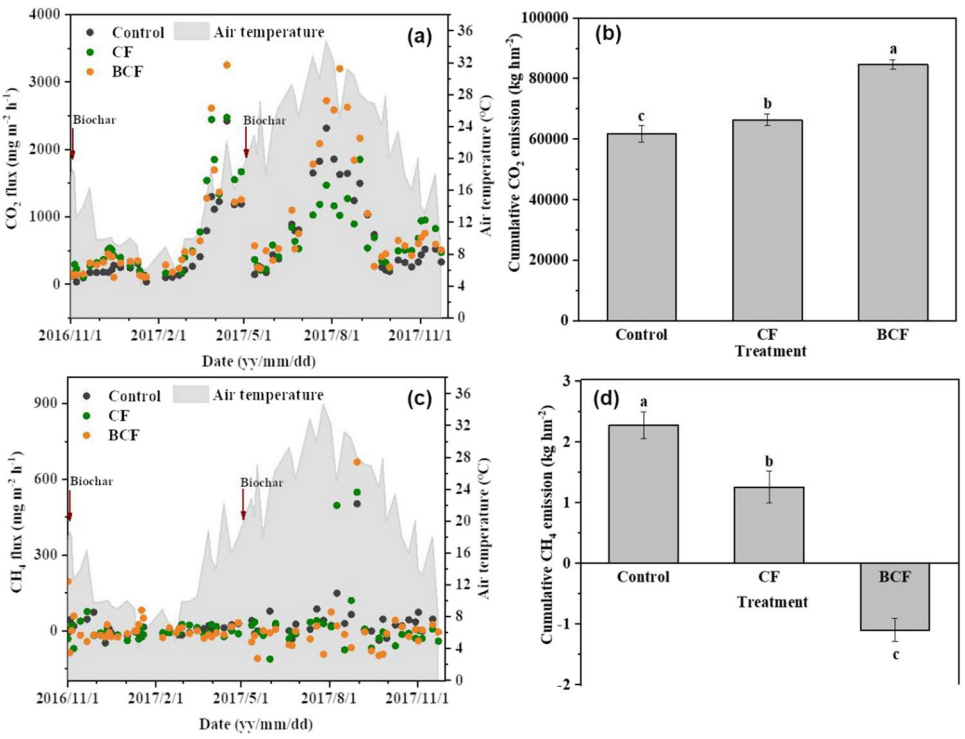

Fig. 1 CO_2 and CH_4 fluxes with time (**a**, **c**) and cumulative CO_2 and CH_4 (**b**, **d**) in different treatments. Control, no fertilizer; CF, chemical fertilizer only; BCF, biochar combined with chemical fertilizer. Different lowercase letters indicate that the differences are significant ($p < 0.05$). Red arrows in scatters indicate the time of biochar application

Results

CO_2 and CH_4 emissions

As shown in Fig. 1a, there were two peaks of CO_2 flux during the experimental period, which were observed in April and August, respectively. The highest CO_2 fluxes with the values of 3254.8 mg m^{-2} h^{-1} and 3201.9 mg m^{-2} h^{-1} were both found in the BCF treatment on April 13 and August 9, respectively. Compared with the control, fertilization (CF and BCF) increased the flux of CO_2, except for the period of higher air temperature (from July to August). Higher CO_2 fluxes were observed in the BCF treatment than in the CF treatment when the air temperature was over 18 ℃. Additionally, the second peak of CO_2 flux in the BCF treatment (on August 9) was later than that in the CF treatment (on July 26). During the experimental period (Fig. 1b), BCF significantly increased the cumulative CO_2 emission by 27.5% and 37.1%, relative to the control and CF treatments, respectively.

In contrast to the CO_2 flux, the variation in the CH_4 flux during the experimental period was not significant (Fig. 1c). However, after the application of biochar, a significant fluctuation in CH_4 flux was observed, especially after the second time of biochar application. Compared with the control, CF and BCF both reduced the cumulative CH_4 emission, and the cumulative CH_4 emission in the BCF treatment was − 1.09 kg hm^{-2} (Fig. 1d).

Temperature sensitivity (Q_{10}) and activation energy (E_a) of CO_2 emission

Because of the negative value of the CH_4 flux, only the temperature sensitivity (Q_{10}) and activation energy (E_a) of CO_2 emission were calculated in this study. The flux of CO_2 has an exponential relationship with the soil temperature (Fig. S1a–c). The dynamic of Q_{10} over time is shown in Fig. 2a. Fertilizer application (CF and BCF) reduced the Q_{10} values during the experimental period. When the first biochar application was applied, BCF reduced the Q_{10} values relative to the CF treatment, but increased the values when the second biochar application was applied. In each season of vegetable growing, the peak of Q_{10} values was observed, especially in April. As shown in Fig. 2b, the lowest value of average Q_{10} was observed in the CF treatment, which was significantly reduced by 29.2% relative to the control. However, there were no significant differences between the CF and BCF treatments, even if a higher value of average Q_{10} ($Q_{10} = 2.1$) was observed in the BCF treatment.

Similar to the Q_{10} dynamic of CO_2 emission, peaks of E_a value were all found in each vegetable growing season, especially in the initial time of vegetable growing (Fig. 2c). Compared with CF, BCF increased the E_a values by 33.7–49.5%, regardless of the number of biochar applications. In addition, the average E_a value in BCF treatment (51.9 kJ mol^{-1}) was

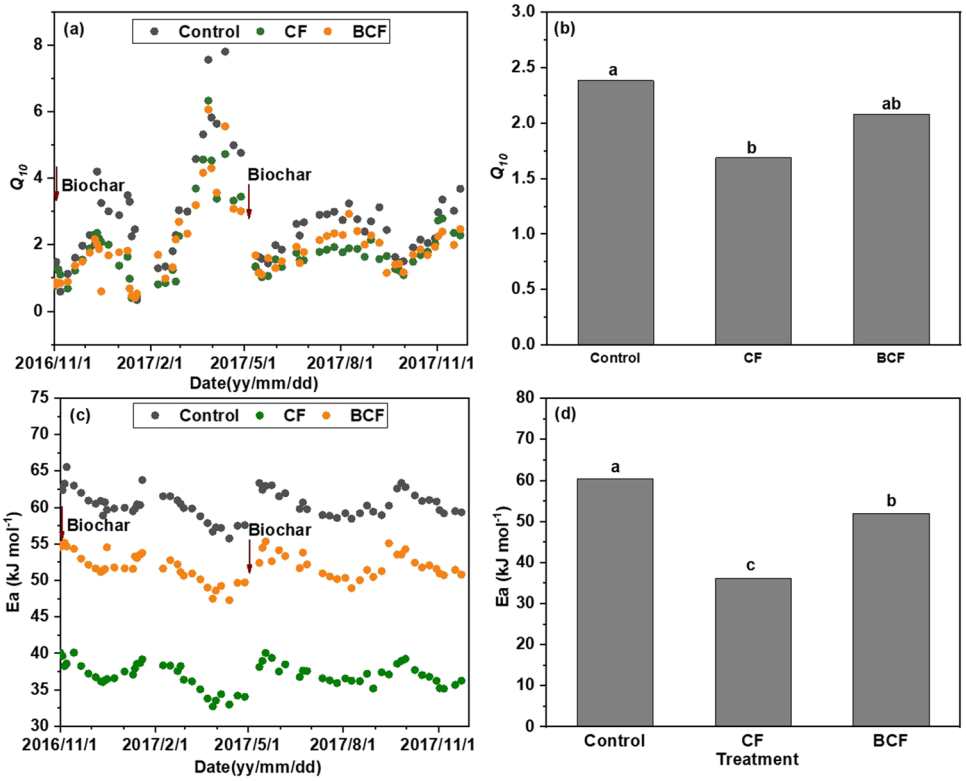

Fig. 2 Temperature sensitivity (Q_{10}) (**a**, **b**) and activation energy (E_a) (**c**, **d**) of CO_2 emissions in different treatments. Control, no fertilizer; CF, chemical fertilizer only; BCF, biochar combined with chemical fertilizer. Different lowercase letters indicate that the differences are significant ($p < 0.05$). Red arrows in scatters indicate the time of biochar application

Table 2 Soil properties in different treatments

	DOC (mg kg^{-1})	MBC (mg kg^{-1})	SOC (g kg^{-1})	pH	WFPS (%)
Control[a]	247.5 ± 86.9a	27.5 ± 8.2a	7.5 ± 0.8b	6.0a	68.1 ± 1.7a
CF[b]	18.3 ± 3.9b	23.0 ± 16.7a	7.8 ± 0.2b	4.6c	48.4 ± 5.0b
BCF[c]	164.5 ± 71.2a	30.7 ± 20.7a	13.1 ± 1.9a	5.0b	53.6 ± 7.6b

Mean ± standard deviation ($n = 3$); different lowercase letters within the same column indicate significant differences ($p < 0.05$)

DOC dissolved organic carbon; *MBC* microbial biomass carbon; *SOC* soil organic carbon; *WFPS* soil water–filled pore space

[a]No fertilizer

[b]Chemical fertilizer only

[c]Biochar combined with chemical fertilizer

significantly higher than those in the control (60.4 kJ mol^{-1}) and CF (36.2 kJ mol^{-1}) treatments (Fig. 2d).

Soil property

Compared with CF, BCF increased the contents of DOC, MBC, and SOC by 800.7% ($p < 0.05$), 33.3% ($p < 0.05$), and 68.9% ($p > 0.05$), respectively (Table 2). In addition, the highest values of soil pH and WFPS were both found in the control, followed by those in the BCF treatment.

Correlation of soil properties, Q_{10}, E_a, and carbon emissions

The first two principal components (PC1 and PC2) accounted for 50.0% and 31.3% of the total variation in PCA, respectively (Fig. 3). The variation in cumulative CO_2 emissions has a positive relationship with SOC but a negative relationship with cumulative CH_4 emissions (Fig. 3). Soil DOC was the key factor influencing the variation in Q_{10} and E_a according to the results of PCA. Correlations among soil properties, Q_{10}, E_a, and carbon emissions (CO_2 and CH_4) are listed in Table S2. The cumulative CO_2 and CH_4 emissions were both significantly associated with SOC ($r = 0.887$ and $r = -0.888$, respectively). The Q_{10} value was correlated with E_a ($r = 0.837$), soil DOC ($r = 0.732$), and pH ($r = 0.765$) ($p < 0.05$ or 0.01). The value of E_a has a significant relationship with soil DOC ($r = 0.933$), pH ($r = 0.873$), and WFPS ($r = 0.792$). In addition, automatic linear modeling revealed that soil SOC, together with MBC, was the primary factor associated with the cumulative CO_2 emissions, as well as SOC and pH associated with the cumulative CH_4 emissions (Fig. 3). Activation energy (E_a) and soil DOC were the key factors influencing Q_{10} and E_a, respectively.

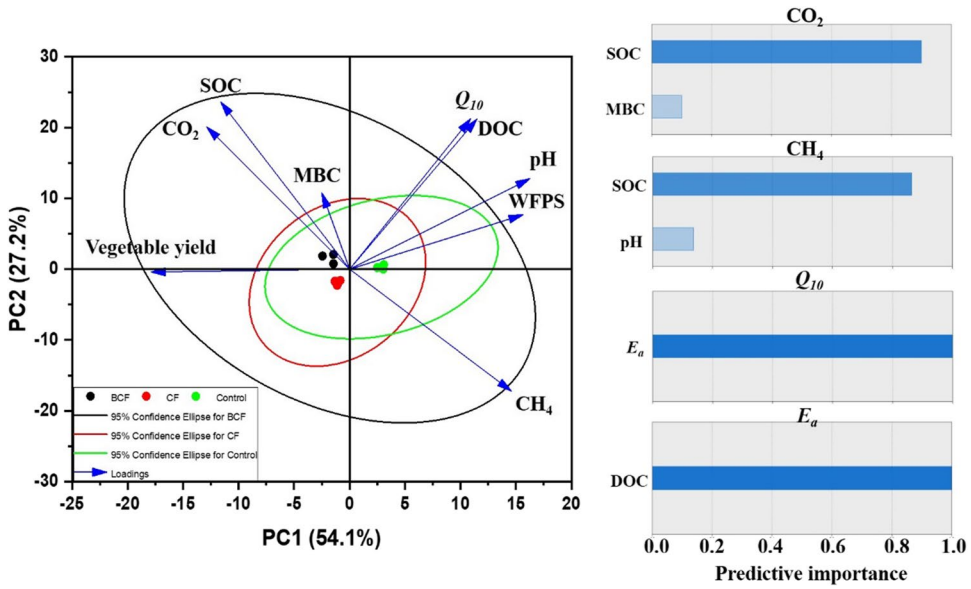

Fig. 3 Principal component analysis (PCA) of soil properties, Q_{10}, and cumulative carbon emissions (left); predictive importance of selected soil properties on cumulative carbon emission Q_{10} and E_a as determined by automatic linear modeling (right)

Discussion

Biochar application influencing the carbon emission

Biochar, as a soil amendment, plays a key role in C utilization and in decreasing greenhouse gas emissions. In general, biochar reduces the CO_2 emissions through the expansion of the soil C pool (Kavitha et al. 2018). In the present study, however, biochar application increased the CO_2 emissions from the soil–plant system during the short-term experiment, relative to the no-biochar (control and CF) treatments (Fig. 1b). The observation of increased cumulative CO_2 emissions in the biochar (BCF) treatment was inconsistent with the previous literature (Lu et al. 2014; Bending et al. 2014; Chen et al. 2017), which demonstrated that biochar application significantly decreased soil CO_2 emissions during short-term incubations. Similarly, the studies of Zhou et al. (2017) and Ge et al. (2020) showed that biochar (produced from bamboo) addition decreased the cumulative soil CO_2 emissions in the field experiments. The inconsistent results may be caused by the different biochar feedstocks, pyrolysis temperatures, and addition rates (Ameloot et al. 2013; Lu et al. 2014; Bending et al. 2014). First, the pyrolysis temperature of 450–500 °C in this study was incomplete oxidization, which may increase volatile matter content and then promote the abiotic release of inorganic C in biochar (Ameloot et al. 2013; Yang et al. 2018). In addition, a greater positive priming effect of biochar was observed immediately at low pyrolysis temperatures (Zimmerman et al. 2011). Second, short-term application of biochar may induce priming effects, causing native soil organic C or labile compounds in biochar to readily decompose by microorganisms (Zimmerman 2010; Wang et al. 2016; Yang et al. 2018). Meanwhile, the combined application of biochar and N fertilization could stimulate CO_2 release from biochar with an increased value of 28.3% (Lu et al. 2014). Third, biochar application in a short period of time provided labile C for soil microbes (especially for the "r-strategist" microbes that are adapted to respond quickly to newly available C sources) and then stimulated soil respiration (Paul and Clark 1989; Zimmerman et al. 2011; Teutscherova et al. 2017; Duan et al. 2020). This hypothesis is supported by the higher contents of soil DOC and MBC in the biochar treatment (Table 2). In addition, the results of automatic linear modeling also verified that the enhanced microbial biomass (e.g., MBC) and C substrates (e.g., SOC) in soils may lead to greater CO_2 emissions (Fig. 3). Although the adsorption and/or encapsulation of biochar can protect native soil labile C from microbial utilization and inhibit the decomposition of native SOC (Zimmerman et al. 2011; Lu et al. 2014; Bending et al. 2014; Chen et al. 2017), the colocation of microorganisms and various nutrients on biochar surfaces and/or in pores may provide a highly suitable habitat for microbes and increase microbial C use efficiency, and subsequently higher CO_2 emissions (Lehmann et al. 2011; Zavalloni et al. 2011). It is worth noting that CO_2 emissions in this study were the net CO_2 emissions from the soil–plant system, which integrated soil respiration, root respiration, and the CO_2 assimilated by plants. The significant negative relationship between total vegetable yield and cumulative CO_2 emissions may indicate the key roles of root respiration and plant photosynthesis in CO_2 emissions (Table S2), especially root respiration. Additionally, biochar application obtained higher total vegetable yields than no-biochar (Table S3). Therefore, short-term biochar and N combined application cannot offset, at least partly, the negative effect of biochar or plant photosynthesis on CO_2 emissions.

It is well known that dryland soil under oxic conditions has the capacity of CH_4 sink due to the soil methanotrophic bacteria oxidizing CH_4 to CO_2 (Suwanwaree and Robertson 2005; Criscuoli et al. 2019). The flux of soil CH_4 is controlled by the production of CH_4 by methanogens and consumption of CH_4 by methanotrophs, as well as the soil conditions that can impact the growth of methanogens and methanotrophs (Le Mer and Roger 2001; Conrad 2007). Consistent with the reported literature (Jeffery et al. 2011; Feng et al. 2012; Qin et al. 2016; Liu et al. 2016b), biochar application in this study significantly reduced the cumulative CH_4 emissions relative to the control and CF treatments (Fig. 1d). A potential explanation is the fact that enhanced soil aeration would increase the activity of methanotrophs due to the biochar's large surface area and pore volume (Wang et al. 2018), which was supported by the negative relationship of cumulative CH_4 emissions and CO_2 emissions (Fig. 3 and Table S2). This result suggested that increased soil CH_4 consumption rather than decreased CH_4 production dominated the influence of biochar in mitigating CH_4 emission from dryland soil–plant system. Another potential explanation, as discussed above, is that the progressive protection of biochar may prevent SOC from being used by methanogens (Zimmerman et al. 2011), resulting in decreased CH_4 production. The higher contents of SOC observed in the BCF treatment may be attributed to the protection of biochar in this study (Table 2). Soil pH plays a key role in affecting both methanogenesis and methanotrophy (Hanson and Hanson 1996; Jeffery et al. 2016). Generally, a pH ranging from 6 to 8 is optimal for most methanogens (Garcia et al. 2000), and high acidity does not favor an increase in the microbial habitability of methanogens (e.g., reducing the abundance of methanogens) (Jeffery et al. 2016). Therefore, a significant increase in CH_4 sink strength was observed in biochar-treated soil with a pH of 5.0, which

is consistent with the findings of Jeffery et al. (2016). However, we observed a CH_4 source in the CF treatment, even if the soil pH was lower than that in the BCF treatment (Table 2). Except for the negative effect of biochar, a possible explanation is that there was more N fertilizer amount in the CF (1200 kg ha^{-1} N fertilizer) treatment than in the BCF (1120 kg ha^{-1} N fertilizer) treatments. The NH_4^+-containing or NH_4^+-delivering fertilizers will compete with CH_4 at the binding sites, consequently decreasing the oxidation of CH_4 (Htun et al. 2017; Huang et al. 2020). Besides, the incorporation of biochar with a high C/N ratio of 142.2 may increase the immobilization of inorganic N (e.g., NH_4^+) and reduce the competitive exclusion of CH_4 (Huang et al. 2020). Meanwhile, in this study, a lower content of NH_4^+ was observed in the BCF (100.7 mg kg^{-1}) treatment than that in the CF (112.3 mg kg^{-1}) treatment. Therefore, short-term application of biochar showed a significant increase in CH_4 sink strength/reduction in CH_4 source strength.

Biochar application influencing the temperature response of CO_2 emissions

In this study, fertilization incorporation reduced the temperature response of CO_2 emissions (expressed as Q_{10} or E_a), compared to the control (Fig. 2a, b). This may be caused by the fact that nutrients (e.g., N and P) from fertilizers changed the substrate C quality, which is linked to soil C emissions (Guo et al. 2017). Previous studies determined that the N addition potentially increased those microbial abundance using labile C and elevated cellulose-decomposing enzyme activity (Carreiro et al. 2000; Keeler et al. 2009). Thus, increased Q_{10} was observed following fertilization or artificial N deposition in previous studies (Liu et al. 2016a; Guo et al. 2017; Ge et al. 2020). The inconsistency of the literature with this study is likely attributed to the different fertilization times (e.g., long-term fertilization (> 10 years) in the study of Guo et al. (2017) and short-term fertilization (approximately 1 year) in this study). Long-term N inputs may change the substrate quality characterized by C complexities and increase the recalcitrant C, leading to an enhanced Q_{10} value (Guo et al. 2017). Generally, the temperature sensitivity of resistant C was higher than that of labile C due to the former needing more activation energy (E_a) and time, according to the enzyme kinetic theory (Davidson and Janssens 2006; Conant et al. 2011). Our observation of the positive relationship between E_a and Q_{10} (Fig. 3 and Table S2) possibly supported the enzyme kinetic hypothesis. Therefore, the reduced Q_{10} under short-term fertilizer inputs may be well explained by the lower E_a in the CF and BCF treatments.

Compared with the CF treatment, biochar addition increased the Q_{10} and E_a, especially after the second application, which is consistent with the report of Wang et al. (2019). Multiple reasons may be responsible for this increase in Q_{10} and E_a. For example, the biochar-induced increase in temperature sensitivity may be attributed to the accumulation of resistant C pools in soil organic matter due to biochar aromatic properties (Zhou et al. 2017; Wang et al. 2019). On the other hand, the increase in Q_{10} and E_a values following biochar application may contribute to enhanced nutrient availability and microbial activities, leading to the decomposition of soil organic matter (Lehmann et al. 2011; Criscuoli et al. 2014), as evidenced by the increased MBC (Table 2), CO_2 flux (Fig. 1a), and N (or P, K) fertilizer utilization efficiency (unpublished data) in the BCF treatment. The increased nutrient availability may reduce the degradability of resistant C, possibly by decreasing the affinity of microbial enzymes (such as phenol oxidase and peroxidase) to substrates (Guo et al. 2017), and thus increase Q_{10} and E_a following biochar application (Fig. 2). In addition, resistant C pools might increase in dry farmland (as in our study) under high microbial activities after biochar addition, contributing to an increase in Q_{10} values (Wang et al. 2019). However, the fact that biochar applications reduced Q_{10} values was also reported in some studies (Pei et al. 2017). These discrepancies may be attributed to the high rate of biochar application in the study of Pei et al. (2017) (i.e., 40–100 t ha^{-1}), which is significantly higher than the rates used in the studies of Zhou et al. (2017) (i.e., 10–30 t ha^{-1}), Kan et al. (2020) (i.e., 1.8–7.2 t ha^{-1}), and our study (i.e., 10–20 t ha^{-1}). More biochar incorporated into soil can increase the non-biochar labile dissolvable C of native soil, which would be entrapped in the porous structure of biochar (Bending et al. 2014). The colocation of microorganisms and entrapped C, as mentioned above, may enhance the availability of soil decomposable C, thus reducing the Q_{10} values (Pei et al. 2017). Although a higher DOC content was observed in soil treated with biochar (Table 2), the low ratio of DOC to SOC (i.e., 1.25%) may indicate that more resistant C remained in soil treated with biochar in the short-term period. Meanwhile, more recalcitrant C with a higher E_a dominated in the soil since the limited labile C was depleted quickly, especially after the second biochar addition.

The temperature response of CO_2 emissions is directly affected by external factors that limit decomposition, except for direct factors (such as substrate availability and microbial enzyme affinity) (Davidson and Janssens 2006; von Lützow and Kögel-Knabner 2009; Fang et al. 2017). Soil pH played a key role in the temperature response of CO_2 emissions in this study due to the significant association of soil pH with Q_{10} and E_a (Table S1 and Fig. 3). Acidifying soil caused by fertilization is characterized by high osmotic pressures, low soil minerals, and high aluminum toxicity, which would reduce microbial activity and consequently decrease the temperature response (Treseder 2008; Liu and Greaver 2010). Thus, the higher soil pH in the BCF treatment may be partly

responsible for the higher temperature response, relative to the CF. In addition, the peak of E_a with time was observed within 1 week of crop transplantation in each growing season, regardless of treatment (Fig. 2c). We speculate that crop cultivation measures may influence E_a possibly by inducing changes in the external and/or direct factors (e.g., root biomass). Unfortunately, the soil indexes with time were not detected in this study. However, the significant relationship of E_a and vegetable yields may indicate the important effect of vegetable cultivation on the temperature response of CO_2 emissions (Table S2). As mentioned above, biochar application may impact CO_2 emissions due to root respiration. Overall, short-term application of biochar increased the temperature response of CO_2 emissions in the soil–plant system.

Conclusion

Short-term application of biochar significantly increased CO_2 emissions from the soil–plant system. However, biochar addition showed a significant reduction in CH_4 source strength in dryland soil, possibly by increasing CH_4 consumption and reducing competition with NH_4^+. Fertilization reduced the temperature sensitivity (Q_{10}) of CO_2 emissions by decreasing the activation energy (E_a). In addition, biochar significantly increased the temperature response (Q_{10} and E_a) of CO_2 emission, relative to solely chemical fertilizer application, which is related to the supplementation of limited labile C and nutrients but highly resistant C following biochar application. External factors (e.g., pH and crop cultivation) play key roles in influencing the change in E_a. Thus, our study suggests that the short-term response of biochar to C gas emissions and temperature should be considered to better understand the long-term effect of biochar on C release and sequestration.

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Data availability The datasets generated and/or analyzed during the current study are available from the corresponding author on reasonable request.

Declarations

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