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Research paper

Biochar decreased the temperature sensitivity of soil carbon decomposition in a paddy field

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ABSTRACT

The effects of biochar, or in general sense the ubiquitously distributed black carbon, on the temperature responses of soil carbon decomposition, is important to our modeling of global carbon balance under climate warming but still poorly understood. In this study, soils of varying biochar contents (0, 31.4% and 53.7% of total soil organic carbon) were collected from a paddy field (C3) that had been amended with corn-cob biochar (C4) for 8 months, and then incubated at 20 °C for 234 days, during which soils were periodically taken out of the incubator and subject to sequentially changing temperatures (cycling between 4 and 28 °C at a step of 4 °C) to determine the temperature sensitivity (estimated by Q_{10}) of soil carbon decomposition. Stable carbon isotopic analysis confirmed that biochar mineralization had become neglectable due to the depletion of labile carbon of biochar in the field. Q_{10} and activation energy (*E*a) of soil carbon decomposition were decreased significantly by biochar amendment, which was particularly evident at the later incubation stages. The biochar-amended soils showed higher respiration rates, potentially mineralizable carbon, and substrate-induced respiration (SIR) than unamended soils. Moreover, following 234 days of incubation, the activities of 7 hydrolytic and one oxidative soil enzymes were all higher in biochar-amended soils, suggesting that biochar increased soil carbon availability and favored microbial activities. There was also more labile carbon (oxidizable by 0.02 M $KMnO_4$) in soils with biochar. In contrast, the microbial metabolic quotient, as estimated by the ratio of basal respiration to SIR, was decreased by biochar. We therefore proposed that the biochar-induced decreases in Q_{10} were not due to enhanced soil carbon protection, but due to the increased amounts of labile carbon (and hence enhanced overall carbon lability) entrapped by biochar as well as the lowered microbial metabolic quotient. Furthermore, it was hypothesized that sorption of water, substrates, enzymes and microorganisms to the "charsphere" (i.e. the biochar-soil interfaces) might decrease the energy costs and thus temperature responses of soil carbon decomposition.

1. Introduction

Black carbon, which refers to a continuum of charred materials generated by incomplete combustion, such as charcoal, soot and graphite, is a highly recalcitrant form of carbon ubiquitous in forest, agricultural soils and sediments (Schmidt and Noack, 2000; Skjemstad et al., 2002; DeLuca and Aplet, 2008). It plays important roles in soil biogeochemistry and ecological processes (Zackrisson et al., 1996; Pietikäinen et al., 2000; Liang et al., 2010). Biochar, a special type of black carbon intentionally produced by pyrolysis of biomass (Sohi et al., 2010), has attracted increasing interests in recent years because of its potentials in locking atmospheric CO_2 for thousands of years (Lehmann, 2007; Kuzuyakov et al., 2009), mitigating climate change and improving soil quality and fertility (Sohi et al., 2010; Bell and Worrall, 2011; Spokas et al., 2012).

Before the widespread application of biochar as a soil additive can be adopted, it is important to thoroughly assess its impacts on soil carbon cycling and biological processes, particularly under scenarios of future climate changes (e.g. climate warming). Generally, the biochar carbon was made up of a labile pool (about 3% of total biochar carbon) and a highly recalcitrant pool (Wang et al., 2016). The labile pool degraded rapidly within the first several months after incorporation into

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soil, while the recalcitrant pool could persist for several millennia (Kuzuyakov et al., 2009; Cross and Sohi, 2011; Luo et al., 2011). The input of biochar carbon to soil can induce changes in the dynamics of native soil carbon (nSC), i.e. the so-called priming effects, which has been found to be highly uncertain (Liang et al., 2010; Singh and Cowie, 2014; Whitman and Lehmann, 2015; Wang et al., 2016; Cui et al., 2017; Luo et al., 2017). Moreover, with the depletion of the labile component of biochar and the occurrence of interactions between biochar and soil minerals, some short-term (i.e. within the first several months following biochar application; e.g. Luo et al., 2011; Cui et al., 2017) effects of biochar on nSC may prove to be transient and different to those in the longer term (Zimmerman et al., 2011; Maestrini et al., 2015).

The responses of soil carbon decomposition to increasing temperature, and the magnitudes of such responses, determine whether soil CO_2 emission will be acceletrated under climate warming, knowledge of which is critical to the modeling and prediction of future climate changes (Davidson and Janssens, 2006; Smith et al., 2008). However, little is known about how biochar amendment would afffect the temperature responses of soil carbon decomposition (Fang et al., 2014, 2015). The temperature sensitivity of soil carbon decomposition (commonly defined as Q_{10}, a factor by which the decomposition rate is multiplied when temperature increases by 10 °C) is kinetically determined by the intrinsic chemical recalcitrance of soil carbon (Davidson and Janssens, 2006), but also complicated by other biotic/abiotic factors including substrate availability (Gershenson et al., 2009), soil carbon stabilization mechanisms (Plante et al., 2010) and microbial physiology (Allison et al., 2010; Conant et al., 2011). Given the profound influences of biochar on soil labile carbon contents (Bell and Worrall, 2011), soil carbon stabilization (Liang et al., 2010), enzymatic activities (Bailey et al., 2010) and microbial communities (Lehmann et al., 2011), it is reasonable to expect that biochar may change the temperature sensitivity of soil carbon decomposition. Understanding this should provide insights into the utility of biomass pyrolysis as a strategy to mitigate climate change (Lehmann, 2007). Such knowledge can also improve our modeling of soil carbon balance in response to climate warming, given the ubiquitous distribution of black carbon in the environment.

While model simulation suggested that inclusion of black carbon in soil carbon pools would lead to prediction of weaker responses of soil carbon release to warming (Lehmann et al., 2008), little empirical evidence yet has been presented to support this. By laboratory incubation of soils added with biochar, Fang et al. (2014) revealed that biochar lowered the Q_{10} of soil carbon decomposition. However, since Fang et al. (2014) incubated soils under constant temperatures, it is unknown whether a different incubation method of Q_{10} determination (e.g. Fang et al., 2005; Hamdi et al., 2013 and references therein) will lead to different conclusions about biochar effects on Q_{10}. Generally, the temperature manipulation method for soil incubation considerably affected estimates of Q_{10} (Hamdi et al., 2013). For instance, when soils were incubated under constant temperatures, the different depletion rates of carbon pools (Conant et al., 2008) and divergent microbial communities (Joergensen et al., 1990) at different temperatures would confound with the temperature responses of soil carbon decomposition.

In this study, we collected soils from a frequently irrigated paddy field that had been amended with concorb-made biochar for 8 months and incubated the soils for another 234 days. We expected that the labile components of biochar carbon had been nearly depleted via microbial consumption or leaching in the field, and thus the transient effects associated with the labile biochar-carbon on soil carbon processes should have become insignificant. This was veryfied by the isotopic analysis of soil-respired CO_2 based on the distinct isotopic signature of biochar (made from the C4 crop *Zea mays*) to that of native soil organic carbon (originating from C3 plants). We periodically determined Q_{10} of soil carbon decomposition during the incubation using a temperature manipulation method by which temperatures were sequentially changed within short time periods (Fang and Moncrieff, 2001; Fang et al., 2005), which could effectively avoid possible confounding effects of factors other than temperature (e.g., substrate depletion or microbial community succession) on Q_{10}. The specific objectives were to (1) test whether biochar amendment changed the temperature responses of soil carbon decomposition, and (2) explore possible mechanisms underlying such effects of biochar in terms of soil carbon availability and microbial functional charactersitics.

2. Materials and methods

2.1. Soil sampling and biochar

A field experiment to assess the impacts of biochar amendment on soil carbon processes was initiated on the Qianshao Farm of the Chongming Island, an island located in the Yangtze Estuary of China. The island had a typical subtropical monsoon climate with a mean annual temperature of 15.3 °C and precipitation of 1003.7 mm (Zhou and Ji, 1989). The farm was reclaimed from estuarine wetlands in the 1960s. Soil mineralogy is dominated by hydromica with minor proportions of kaoline, chlorite and vermiculite (He, 1992).

Biochar used in the field experiment was purchased from the Mingyan group co., LTD., Wuxi, China. It was produced from corn (*Zea mays*, a C4 plant) cobs by fast pyrolysis at 350–550 °C. Before incorporation to soil, the corn-cob biochar was ground by a shredding machine, passed through a 2-mm sieve and thoroughly homogenized. The relevant properties of biochar were pH 9.63 ± 0.13, total carbon 57.2 ± 0.28%, total nitrogen 0.88 ± 0.03%, and stable carbon isotopic signature ($\delta^{13}C$) −13.28 ± 0.32‰.

In June 2014, a paddy field that had been grew with C3 crops (rice and wheat/barley) since reclamation was selected and biochar was amended at rates of 0, 40 and 100 t ha^{-1}, using a randomized complete block design with four replicate plots (4 × 5 m^2) per treatment. The distinct stable carbon isotopic signature of biochar (C4) and soil organic matter (C3) would allow partitioning of carbon fluxes from the two sources (e.g. Luo et al., 2011). The 100 t ha^{-1} biochar application rate was extraordinarily high in practice, but was used only to strengthen biochar effects on soil (following Major et al., 2010). Biochar was spread evenly on soil surface and incorporated to the depth of 15 cm by ploughing with a rotocultivator. Immediately following this, the field was flooded with water for rice (*Oryza sativa*) cultivation. The paddy field was irrigated/drained at least once every week at the early and medium stages of rice growth, but at lower frequencies before rice maturation. It was observed that the surface runoff accompanying the frequent irrigation/drainage and the high summer rainfall caused substantial loss of biochar. Calculation based on ^{13}C isotopic analysis revealed that only around 20% of the applied biochar remained in the field after rice cultivation (Pei et al., 2016). Rice was harvested in October, following which winter wheat (*Triticum aestivum*) was grown.

Soils used for laboratory incubation were collected from the paddy field about 8 months after biochar amendment, i.e. during the wheat growth period. Because the frequent irrigation/drainage and other management practices caused large heterogeneity of biochar distribution in the field, sampling was only conducted at locations where biochar was clearly visible at soil surface. For each treatment, soils were sampled to 15 cm from multiple locations and combined as a composite sample (about 5 kg in weight), which was repeated three times to get three composite samples per biochar amendment rate. Prior to the incubation, we quantified biochar contents in the collected soils based on their stable carbon isotopic composition (Table 1).

2.2. Laboratory incubation

Treatments in laboratory soil incubation were termed as CK, BL (low biochar content) and BH (high biochar content), corresponding to treatments initially receiving 0, 40 and 100 t ha^{-1} biochar in the field. Soils were handpicked to remove visible roots or animals, sieved to <