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Plant-biochar interactions drive the negative priming of soil organic carbon in an annual ryegrass field system

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ABSTRACT

There is a knowledge gap on biochar carbon (C) longevity and its priming effects on soil organic carbon (SOC) and recent root-derived C under field conditions. This knowledge would allow the potential of biochar in long-term soil C sequestration to be established. However, most studies on biochar C longevity and its priming effect have been undertaken in plant-free laboratory incubations.

A 388 d field study was carried out in the presence of an annual ryegrass (C_3) growing on a rhodic ferralsol with established C_3/C_4 plant-derived SOC ($\delta^{13}C$: −20.2‰) in a subtropical climate. A ^{13}C-depleted hardwood biochar ($\delta^{13}C$: −35.7‰, produced at 450 °C) was applied at 0 and 30 dry t ha^{-1} and mixed into the top 100-mm soil profile (equivalent to 3% w/w). We report on the differentiation and quantification of root respiration and mineralisation of soil-C and biochar-C in the field. Periodic $^{13}CO_2$ pulse labelling was applied to enrich $\delta^{13}C$ of root respiration during two separate winter campaigns ($\delta^{13}C$: 151.5–184.6‰) and one summer campaign ($\delta^{13}C$: 19.8–31.5‰). Combined soil plus root respiration was separated from leaf respiration using a novel in-field respiration collar. A two-pool isotope mixing model was applied to partition three C sources (i.e. root, biochar and soil). Three scenarios were used to assess the sensitivity associated with the C source partitioning in the planted systems: 1) extreme positive priming of recent SOC derived from the current ryegrass (C_3) pasture; 2) equivalent magnitude of priming of SOC and labile root C; and 3) extreme positive priming of the native C4-dominant SOC.

We showed that biochar induced a significant negative priming of SOC in the presence of growing plants but no net priming was observed in the unplanted soil. We also demonstrated the importance of experimental timeframe in capturing the transient nature of biochar-induced priming, from positive (day 0–62) to negative (day 62–388). The presence/absence of plants had no impact on biochar-C mineralisation in this ferralsol during the measurement period. Based on a two-pool exponential model, the mean residence time (MRT) of biochar varied from 351 to 449 years in the intensive pasture system to 415–484 years in the unplanted soils.

1. Introduction

Over the past decade, a growing body of knowledge on biochar C longevity in soil has accumulated from many laboratory studies (Cross and Sohi, 2011; Jones et al., 2011; Keith et al., 2011; Zimmerman et al., 2011; Kuzyakov et al., 2014; Singh and Cowie, 2014). Two major C pools within biochar have been proposed: labile and recalcitrant C (Zimmerman et al., 2011). The labile C pool breaks down readily with observed half-lives in the laboratory ranging from 27 d (wood biochar 400 °C) to 438 d (grass biochar 650 °C), whereas the recalcitrant C pool tends to be persistent with observed half-lives from 114 years (oak biochar 250 °C) to 1120 years (wood biochar 550 °C) (Singh et al., 2012; Zimmerman and Gao, 2013). Although biochar can be chemically and physically recalcitrant, it is not inert in soil (Lehmann and Joseph, 2009). Based on a limited number of field studies on biochar C longevity,

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the half-lives of biochar can vary from 3.3 years (Nguyen et al., 2008) to millennia (Zimmerman and Gao, 2013). However, the biochar used in most of these field studies were produced by natural fires or slash-and-burn agricultural systems. Exceptionally, Major et al. (2010) estimated the half-life of the wood biochar (produced between 400 and 600 °C) used in their field study to be around 42 years.

The change in the mineralisation rates of soil organic carbon (SOC) by soil treatments is defined as priming effect (Blagodatskaya and Kuzyakov, 2008). The direction of priming can be either positive or negative leading to increased or lowered SOC mineralisation, or changing from one to the other over time. The challenge in investigating biochar-SOC interaction is the slow turnover of the recalcitrant biochar-C pool, as changes are too small to detect over a typical experimental timeframe (Zimmerman et al., 2011; Singh et al., 2012). In general, the majority of published incubation studies on biochar-induced priming effects are less than 90 d. Most of these studies showed single-direction priming. In a 14 d incubation study, Cross and Sohi (2011) demonstrated positive priming of 6.1–25.1% after application of sugarcane bagasse biochars (350 °C and 550 °C respectively), whereas Lu et al. (2014) found that biochar lowered the mineralisation of native SOC by up to 68.8% within 30 d. Contrasting these results, the medium-term studies, ranging from 90 d (Zimmerman et al., 2011) to 730 d (Major et al., 2010), showed the transient nature of biochar-induced priming effects. In a 158 d incubation study (Maestrini et al., 2014), a ryegrass biochar (450 °C) caused positive priming of SOC in the first 18 days then negative priming over the rest of the experiment. In contrast, in a 5 year long laboratory incubation study, Singh and Cowie (2014) reported that biochar maintained positive priming, dominantly in the first 2.3 years, with a decreasing magnitude from 2.3 to 5 years.

Few laboratory incubation studies have considered key processes likely to control SOC priming in the field (Whitman et al., 2015). Nguyen and Lehmann (2009) pointed out wetting and drying cycles can increase carboxylic and hydroxyl functional groups in biochar. Repeated wet–dry cycles can also lead to SOC losses via breakdown and reformation of macro-aggregates (Denef et al., 2001), and may alter microbial community structure which is responsible for SOC turnover (Jastrow et al., 2007). Fang et al. (2015) showed elevated temperatures can shorten the mean residence time (MRT) of low temperature (450 °C) biochar. The few existing published biochar field studies have generally only measured total C mineralised from biochar-soil mixtures (Novak et al., 2010; Scheer et al., 2011), or observed changes in soil C content (Major et al., 2010; Slavich et al., 2013) but have not determined the relative impact of biochar on mineralisation of existing SOC and recent C from roots and their exudates. These aboveground consequences of belowground interactions are critical for mechanistic understanding behind these in-field observations (Wardle et al., 2004). In a forest soil study, Singh et al. (2014) suggested that pyrogenic organic matter (PyOM) may sorb the added root-derived labile C and thereby enhance organo-mineral interactions on oxidised biochar surfaces. Biochar-induced negative priming has only been demonstrated recently in the presence of plants using the trenching method (Ventura et al., 2014) or ^{13}C-labelled PyOM (Whitman et al., 2014). However, in both studies root-C contributions to total CO_2–C emissions were not reported.

Understanding biochar C longevity in field soil in the presence of plants and its interaction with SOC and recent root-derived C is essential for assessing its role in long-term C sequestration. However, there is a knowledge gap on the assessment of biochar longevity, its mean residence time (MRT) and its priming of SOC in the presence of an annual ryegrass under field conditions. To address this, we aimed to quantify the influence of biochar on the *in-situ* turnover of soil and root-derived carbon (i.e. priming effect). We partitioned and quantified C mineralisation from a ^{13}C-depleted *Eucalyptus saligna* biochar (450 °C) in a subtropical pasture on a rhodic ferralsol. We also estimated the MRT of biochar both in the presence and absence of plants in the field.

2. Material and methods

2.1. Biochar and site characteristics

Biochar was produced by slow pyrolysis (5–10 °C min^{-1}, at a highest treatment temperature of 450 °C) with a residence time of 40 min using a batch reactor at Pacific Pyrolysis (Somersby, NSW, Australia). The feedstock was derived from biomass of *E. saligna* ($\delta^{13}C = -35.6‰$), which has grown in an elevated, ^{13}C-depleted CO_2 environment for two years (Barton et al., 2010). The feedstock comprised mainly woody stem (~80% of total biomass), plus small twigs and leaves. The analysis and physicochemical properties of the biochar are detailed in Singh et al. (under review). The pH of the biochar was 9.8, measured in distilled water in a w/w ratio of 1:5. Total C and total N were 66.8 mg kg^{-1} and 1.04 mg kg^{-1} respectively. The H/C_{org} ratio was 0.63%. Cation Exchange Capacity (CEC) was 12 $mmol_c$ kg^{-1} (Table 1). The biochar was air-dried and sieved to below 2 mm.

The subtropical field site was located at the Wollongbar Primary Industries Institute (28°49′S, 153°23′E, elevation: 140 m), Wollongbar, New South Wales, Australia. This site was converted from subtropical forest to dairy pasture in 1894 (Kirton, personal communication). The site has been managed as an intensive diary pasture on the experimental station since then. The detailed soil analysis for the 0–100 mm layer and classification is provided in van Zwieten et al. (2015). In brief, the soil is fine-textured and iron-rich with kaolinite, gibbsite and goethite mineralogy. The field experiment was conducted on a mixed C_3/C_4 vegetated rhodic ferralsol, characterised by highly permeability (Isbell, 1996). The soil was acidic (pH 4.5; 1:5 $CaCl_2$) with 4.5% total carbon content; total Fe 8.4% and total Al 6.7% (van Zwieten et al., 2010a).

2.2. Field set-up

Twelve circular microplots (440 mm in diameter) were established based on a design by Major et al. (2010), with a boundary of heavy duty polyethylene (PE) barrier (150 mm into soil). These were arranged in a 6 × 2 array at 2 m aside and treatments were allocated randomly in a complete block design (Fig. 1, top). The treatments were unamended (control) or ^{13}C-depleted wood biochar amended at 30 t ha^{-1} to the 0–100 mm depth (equivalent to 3% w/w, biochar-amended) (Fig. 1). This biochar dose falls within a common range of biochar application dose in Australia (Chan et al., 2008; van Zwieten et al., 2010a). Six replicates were established per treatment. The site was prepared by mowing existing pasture to remove most of the aboveground biomass. To ensure uniformity between microplots, the top 100 mm of soil was carefully excavated from all microplots, air-dried and manually sieved to less than 2 mm. No stones were found in the soil. Hence the unamended soil was repacked to approximately the same field bulk density of 1.01 g cm^{-3}. The bulk density of biochar was determined to be 0.331 g cm^{-1} using method described in Quin et al. (2014). The weight of soil/biochar mixture in the top 100 mm profile was then calculated based on the bulk density of biochar and soil at 3% w/w mixing ratio. The soil/biochar mixture was carefully packed into respiration collars using similar force as to that in the unamended plots. There was surplus soil remaining in biochar-amended soils due to the decrease in bulk density by biochar additions.