Ecological Engineering 94 (2016) 329–336

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Effects of biochar on carbon mineralization of coastal wetland soils in the Yellow River Delta, China

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ARTICLE INFO

Article history:
Received 22 June 2015
Received in revised form 1 June 2016
Accepted 2 June 2016
Available online 18 June 2016

Keywords:
Biochar
Coastal soil
Carbon mineralization
Microbial biomass carbon
Coastal wetlands
the Yellow River Delta

ABSTRACT

Biochar (BC) application for improving soil quality and carbon sequestration has generated great interest for scientists and policy makers. BC can influence soil organic carbon (SOC) mineralization through priming effects. Positive, negative or no priming effects on C mineralization has been observed following BC additions to soils. However, uncertainty still remains about the influence of biochar on SOC mineralization in the coastal wetland soils of the Yellow River Delta, China. Therefore, a five months of incubation experiment using the coastal wetland soil was conducted to investigate the effects of adding BC produced from peanut shell at 350 °C on SOC mineralization at the rates of 0% (0%BC), 0.1% (0.1%BC), 1% (1%BC) and 3% (3%BC) (w/w). BC addition increased the cumulative CO_2 emissions, indicating that the cumulative SOC mineralization was enhanced in the coastal soil by BC application. However, the increased C mineralized only accounted for 1.71%, 0.32% and 0.17% of the BC-C added in the 0.1%BC, 1%BC and 3%BC treatments, respectively. Moreover, the experimental values of SOC mineralization was much lower than that of the theoretical values in both 1%BC and 3%BC treatments, indicating that a negative priming effect occurred. This may be explained by two reasons: (1) the conversion process of SOC to dissolved inorganic C (DIC) was accelerated by BC addition, which was confirmed by SEM image; (2) the amounts of available C substrate and microorganisms decreased via the sorption of labile organic C (LOC) and microorganisms onto BC. These results suggest that BC application will enhance soil C storage in the salinized wetland soils.

1. Introduction

Biochar (BC) is a fine-grained and porous carbon-rich material, which is produced from biomass pyrolysis in the partial or total absence of oxygen (Sohi et al., 2010). BC addition into soils could improve soil quality (e.g., bulk density, porosity, water and nutrient retention) (Jones et al., 2012) and thus may enhance crop growth (Zheng et al., 2013). Meanwhile, BC has been regarded as a promising material for sequestrating carbon (C) due to its chemical and biological stability (Pereira et al., 2011; Singh et al., 2012). Coastal wetlands (e.g., salt marshes, mangrove), the interfaces between terrestrial land ocean ecosystems, play an important role in the global C cycle by acting as they withdraw CO_2 from the atmosphere and store it in living plant tissue (Mcleod et al., 2011; Pendleton et al., 2012). Globally, at least 430 Tg of C is sequestrated in the upper 50 cm of salt marsh soils (Chmura et al., 2003). Unfortunately, it was reported that the wetlands had seriously degraded and about 50% of the global wetlands had been exploited for agricultural and other land uses (Huang et al., 2012; Verhoeven and Setter, 2010), resulting in a significant reduction of the wetland C stocks. For example, more than half (approximately 7082.0 km^2) of salt marshes have been reclaimed for other land uses in China. Therefore, a cost-effective and sustainable soil amendment with multiple benefits for improving soil quality, restoring primary productivity, conserving ecological functions and thus increasing soil C stock is urgently required. BC has currently generated considerable interests for scientists and policy makers, may be a potential solution for remediating these degraded wetlands and enhancing C sink due to its multiple benefits in soils (Sun et al., 2014).

Globally, for the wetland ecosystem, only accounting for 4–6% of the terrestrial ecosystem, the C stocks accounts for 12–20% of the

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http://dx.doi.org/10.1016/j.ecoleng.2016.06.004
0925-8574/

terrestrial carbon reservoirs, and is higher than those in agricultural ecosystem and temperate forest ecosystem (Fennessy, 2014). Thus the mineralization of soil organic carbon (SOC) in the coastal soils has been received much interest because the minor change of C pool will have a remarkable effect on the balance of global C cycle and the global climate change (Chambers et al., 2013; Heimann and Reichstein, 2008). Positive, negative or no priming effects of BC additions on SOC mineralization in agricultural and forest soils have been reported (Cross and Sohi, 2011; Lu et al., 2014; Luo et al., 2011; Whitman et al., 2014; Zimmerman et al., 2011). It is widely believed that the positive priming effects are caused by the increased soil microbial biomass and activities via supporting habitat and labile C source because of BC additions (Farrell et al., 2013; Luo et al., 2011), the enhanced production of soil extracellular enzymes owing to the co-metabolized between BC and SOC (Farrell et al., 2013; Kuzyakov et al., 2000). Negative priming was frequently attributed to inhibition of microbial growth and activity because of toxicant substance in BCs (e.g., PAHs, dioxins) (Deenik et al., 2010; Zimmerman et al., 2011), and direct sorption of labile organic matter and nutrients (e.g., NH_4^+ and PO_4^{3-}) onto BCs (Dempster et al., 2012; Lu et al., 2014). These different responses of SOC mineralization in the BC-amended soils resulted from the complicated properties of BCs and various types of soils, as well as the complicated interactions between BCs and soils. Thus, uncertainty still remains about the influence of biochar on SOC mineralization in different types of soil.

Soils in coastal wetland are often alkaline and of low fertility, but accumulate organic matter because of relatively low rate of organic matter decomposition (Han et al., 2014; Huang et al., 2012; Wang et al., 2012), and more than 90% of total C (TC) can be stored in wetland soils (Chen et al., 2013). The Yellow River Delta, one of the most active regions of land-ocean interaction in the world, is a piece of young land that was formed less than 150 years ago (Han et al., 2014). In the Yellow River, $\sim 10.5 \times 10^{11}$ kg of sediment are transported annually and most of them were deposited at the river mouth forming new wetlands (Cui et al., 2009). As a newly formed estuarine delta, the soil is naturally characterized by extensive coverage of primary salinization, accounting for 47–70% of the land area of the Yellow River Delta (Xie et al., 2011). In addition, the surface soils may experience large fluctuations between freshwater and seawater, as well as between groundwater and surface water, which have the potential to significantly alter the C mineralization rate, microbial activity and nutrient dynamics (Wang et al., 2016). Moreover, the pool of soil inorganic carbon (SIC), formed in soil by precipitation of Ca^{2+} from soil parent material with dissolved CO_2, comprising approximately 38% of the total soil C pool, was always ignored in evaluating the soil amendments on C cycling (Gocke et al., 2011). Therefore, due to the complicate soil conditions in the Yellow River Delta, many questions remain about how wetlands might mitigate (or contribute to) global warming as a function of C sequestration, thus hindering to evaluate C sequestration potential in the wetland ecosystems. It has been reported that BC application greatly reduced pH, increased SOC content and cation exchange capacity (CEC), and enhanced available phosphorus (P) of the coastal soil in the Yellow River Delta (Wu et al., 2014). Sun et al. (2014) found that the respiration rate was much lower in the saline soil amended with wheat straw biochar compared to the uncharred wheat straw, but the mechanism was not clear. Song et al. (2011) first reported that adding biochar increased the abundance and altered the composition of ammonia oxidizers. However, these findings including BC effects on C mineralization are not sufficient to evaluate C sequestration potential in the Yellow River Delta. Clearly, a thorough understanding and quantification of the adding BC effects on SOC mineralization are important to assessing the true C sequestration potential and climate change mitigation value of BC incorporation to the coastal soil. Therefore, the following questions were investigated in the present study: (1) Is there a consistent negative priming response to the peanut shell BC addition into a coastal saline soil collected from the wetlands in the Yellow River Delta? (2) Is there a dose-response effect for the priming effect? (3) How dose adding BC affect SIC in C mineralization? To address these questions, an incubation experiment over five months was conducted to investigate the effects of adding BC produced from peanut at 350 °C on SOC mineralization in a coastal saline soil collected from the Yellow River Delta wetlands. The specific study aimed to (1) determine the effects of BC addition with different rates on SOC mineralization, (2) investigate the effects of adding BC on SIC content and (3) discuss the underlying mechanisms responsible for C cycling in the BC-amended soil.

2. Materials and methods

2.1. Soil sampling and analyses

The surface soil (0–20 cm) was collected from the wetlands in the Yellow River Delta (37°49′31.3″N, 118°59′47.2″E) in Dongying, China. The vegetation in the sampling site was dominant with reed (*Phragmites australis*). The soil samples were air-dried and the large plant debris and roots were removed, then homogenized and sieved to pass a 2-mm sieve. The soil was stored in glass jars at 4 °C for further analysis.

The content of total organic carbon (TOC) was measured by an elemental analyzer (FLASH-2000, Thermo, America). The content of dissolved organic C (DOC) and dissolved inorganic C (DIC) were determined in 0.50 M K_2SO_4 (1:5, w/v) using an organic C (OC) autoanalyzer (TOC-Vario, Elementar, Germany). Microbial biomass carbon (MBC) was measured using fumigation extraction method (Wu et al., 1990). Soil pH was measured in the deionized water at a ratio of 1:2.5 (w/v). The soluble salt content was obtained by a gravimetric method. Bulk density and water holding capacity (WHC) were measured using cutting ring method. Cation exchange capacity (CEC) was determined using the sodium acetate-flame photometry method. The selected physico-chemical properties of the soil are: pH 8.09, TC 1.13%, TN 0.03%, C:N ratio 34, CEC 9.77 cmol/kg, EC 1548 μS/cm, NH_4^+-N 4.58 mg/kg and NO_3^--N 1.41 mg/kg.

2.2. BC production and characterization

BC was produced from peanut shell at 350 °C for 2 h using a vacuum tube furnace (O-KTF1200, China) under a N_2 flow of 500 mL min^{-1} (Zheng et al., 2013). The samples were milled to pass a 2-mm sieve and homogenized for further analysis. The pH values of the samples were measured at a ratio of 1:20 (w/v) in deionized water after being shaken for 24 h at 130 rpm. Total C (TC) and total N (TN) were determined using an elemental analyzer (FLASH-2000, Thermo, USA). The content of water soluble C (including DOC and DIC) was measured using the extracting method with deionized water at the ratio of 10:1 (water/biochar, v/w) (Luo et al., 2011). The spectra of Fourier Transform Infrared (FTIR) of KBr pellets prepared with 0.5% BC were recorded (Tensor 27, Bruker Optics, Germany) with a diffuse-reflectance sampling accessory between 450 and 4000 cm^{-1} with one hundred scans averaged with a resolution of 4 cm^{-1} (Fig. S1). The chemical properties of the BC are: pH 8.54 ± 0.01, TC 440 ± 0.04 g/kg, TN 10.1 ± 0.04 g/kg, CEC 15.2 ± 0.01 cmol/kg.

2.3. Incubation experiment

Soil incubation experiments were conducted in 250-mL Erlenmeyer flasks filled with 40 g pre-incubated soil amended with BC at the rates of 0%, 0.1%, 1% and 3% (w/w), thereafter referred to as 0%BC, 0.1%BC, 1%BC and 3%BC, respectively. The soils were incubated at