Soil Biology & Biochemistry 72 (2014) 152–162

Contents lists available at ScienceDirect

Soil Biology & Biochemistry

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

Medium-term effects of corn biochar addition on soil biota activities and functions in a temperate soil cropped to corn

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ARTICLE INFO

Article history:
Received 22 October 2013
Received in revised form
19 November 2013
Accepted 29 January 2014
Available online 11 February 2014

Keywords:
Biochar
Microorganisms
Soil fauna
Litter
Decomposition
Mineralization

ABSTRACT

Biochar addition to soil has been generally associated with crop yield increases observed in some soils, and increased nutrient availability is one of the mechanisms proposed. Any impact of biochar on soil organisms can potentially translate to changes in nutrient availability and crop productivity, possibly explaining some of the beneficial and detrimental yield effects reported in literature. Therefore, the main aim of this study was to assess the medium-term impact of biochar addition on microbial and faunal activities in a temperate soil cropped to corn and the consequences for their main functions, litter decomposition and mineralization. Biochar was added to a corn field at rates of 0, 3, 12, 30 tons ha^{-1} three years prior to this study, in comparison to an annual application of 1 t ha^{-1}.

Biochar application increased microbial abundance, which nearly doubled at the highest addition rate, while mesofauna activity, and litter decomposition facilitated by mesofauna were not increased significantly but were positively influenced by biochar addition when these responses were modeled, and in the last case directly and positively associated to the higher microbial abundance. In addition, in short-term laboratory experiments after the addition of litter, biochar presence increased $NO_2 + NO_3$ mineralization, and decreased that of SO_4 and Cl. However, those nutrient effects were not shown to be of concern at the field scale, where only some significant increases in SOC, pH, Cl and PO_4 were observed.

Therefore, no negative impacts in the soil biota activities and functions assessed were observed for the tested alkaline biochar after three years of the application, although this trend needs to be verified for other soil and biochar types.

1. Introduction

Biochar is a carbon(C) -rich product obtained by thermal decomposition of biomass at relatively low temperatures (<700 °C) and low oxygen concentration, in a process known as pyrolysis. During this process heat, flammable gases and liquids are produced together with a solid residue, biochar. The process resembles traditional charcoal production, but biochar is used as a soil amendment and not for energy generation (Lehmann and Joseph, 2009). More recently, biochar has been more narrowly defined in terms of its capacity to sequester C and improve soil functions (Verheijen et al., 2010). Due to its particulate nature and its chemical structure, biochar is more stable than any other organic amendment which provides high recalcitrance to microbial decomposition (Spokas, 2010), which has led to the consideration of biochar production as a C-negative technology for climate change mitigation (Woolf et al., 2010). Biochar application to soil and knowledge of its benefits to improve soil fertility is not new and has been practiced in traditional agriculture in many regions (Ogawa and Okimori, 2010). However, the recent activity in biochar research and development has generated broad interest that has lead to a rapid spread of the technology.

Biochar is able to improve soil fertility in some soils (Verheijen et al., 2010; Jeffery et al., 2011; Kookana et al., 2011; Spokas et al., 2012; Biederman and Harpole, 2013) as a result of its effects on physico-chemical and biological properties. Biochar has been shown to improve water retention, aggregation and permeability in some soils (Downie et al., 2009; Busscher et al., 2010; Liu et al., 2012), or increase the pH of acid soil (Jeffery et al., 2011), as well

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http://dx.doi.org/10.1016/j.soilbio.2014.01.035
0038-0717/

as increase plant nutrient availability in nutrient-limited agroecosystems (Major et al., 2010). Various mechanisms have been suggested for the latter such as: (1) the initial addition of soluble nutrients contained in the biochar (Sohi et al., 2010) and the mineralization of the labile fraction of biochar containing organically bound nutrients (Lehmann et al., 2009); (2) reduced nutrient leaching due to biochars' high cation exchange capacity (Liang et al., 2006; Cheng et al., 2008; Laird et al., 2010; Spokas et al., 2012); (3) lower gaseous N losses by ammonia volatilization (Taghizadeh-Toosi et al., 2012) and N_2 and N_2O by denitrification (Cayuela et al., 2013); and (4) a retention of N, P and S associated with the increase in biological activities and/or community shifts (Pietikäinen et al., 2000; Thies and Rillig, 2009; Lehmann et al., 2011; Güereña et al., 2013). Some of these mechanisms involve soil biota, and this is why effects on soil fauna might translate into changes in nutrient availability (Altieri, 1999, Lavelle et al., 2006). Despite this fact, effects on soil biota are one of the most understudied topics in biochar research (Lehmann et al., 2011), and many of the observed effects may be explainable with changes in soil biota.

In agroecosystems decomposer microorganisms are essential for nutrient release from soil organic matter to sustain crop production in addition to the inputs of fertilizers (Bardgett, 2005). If biochar causes shifts in microbial communities, C cycling can also be affected (Nielsen et al., 2011), as well as other nutrients, and influence primary production or the fauna relying on microbiota. Not only changes in microorganism activity, but that of any soil biota group may have effects on other groups due to the complexity of below-ground food webs (Bardgett, 2005). Therefore, an understanding of biochar effects on the interaction between a range of soil biota groups is needed.

Research on the effects of biochar on soil biota has been largely restricted to soil microbial abundance and activity. The change of the physicochemical environment, such as increased water and nutrient retention, and the provision of a refuge habitat protecting microorganisms from predators have been proposed as mechanisms (Lehmann et al., 2011; Ennis et al., 2012). However, studies on the impact on other biological groups are scarce in the scientific literature, especially with respect to soil fauna (Lehmann et al., 2011). In addition, the consequences of such impacts on soil functions such as decomposition and mineralization are poorly understood. It has been hypothesized that biochar might positively affect soil biota through the increase in soil aggregation and porosity, pH, moisture retention and soil temperature, as well as nutrient retention (McCormack et al., 2013), although negative effects might be also be expected with an enhanced retention of toxic substances, such as ammonium and pesticides (Ennis et al., 2012; McCormack et al., 2013), and the release of pollutants from biochar, such as pyrolysis oils (Gell et al., 2011) and PAH (Hale et al., 2012). Currently there is a need for demonstration of the environmental benefits of biochar while avoiding detrimental effects on environmental health (Verheijen et al., 2010). Some biochars might pose a direct risk to soil biota and their functions (Liesch et al., 2010; Weyers and Spokas, 2011), and may explain some of the negative crop yields reported in literature (Spokas et al., 2012).

The aim of our study is assessing the medium-term effects of biochar additions on microbial and faunal activity and their main soil functions, decomposition and mineralization.

2. Methods

2.1. Experimental plots

The experimental plots were located at Cornell University's Musgrave Research Farm in Aurora, NY, USA (42°43′48.64″N, 76°39′16.03″W), continuously cropped to corn for more than 30 years in a soil and with an experimental design described in detail by Güereña et al. (2013). The experimental site was divided into plots of 4.5 × 7.5 m (33.7 m^2), with a 2-m buffer strip between them. Three plots were prepared per biochar addition rate in a completely randomized design. In April 2007, biochar was applied before planting, at rates of 0, 3, 12, 30 t ha^{-1}. In addition, an annual application of 1 t ha^{-1} was tested using the same batch of biochar (applied in 2007, 2008, and 2010, but not in 2009). Biochar was incorporated to plots by hand rake and shovel to a depth of approximately 50 mm which was then followed by mechanical tillage to about 0.13 m uniformly for all treatments.

The biochar was produced from corn stover by slow pyrolysis (30 min, 600 °C) at BEST Energies Inc. (Somersby, Australia), and its properties are described in Güereña et al. (2013). The ecotoxicological characterization of this biochar demonstrated no inhibition for the reproduction of soil collembolans (ISO, 1999) and enchytraeids (ISO, 2004) in soil-fresh biochar mixtures (0.2–14%, w/w) after 28 d of exposure (data no shown).

In the 2010 growing season of this study, three years after the application of biochar, a NPK fertilizer (10-20-20) was applied at planting (mid-May) at a rate of 12.3 kg N ha^{-1}. Three weeks after planting (early July), a secondary fertilization was applied at rates of 100.8 kg N ha^{-1} (corresponding to 90% of the recommended N application rate).

Plots were sown with a maize crop (Pioneer Hybrid 38M60 Triple stack, Pioneer Hi-Bred International, Inc., Johnston, IA, USA), at a rate of 79,287 seeds ha^{-1}. No pesticides were applied that year with the exception of pre-emergence herbicides applied just after sowing (atrazine and Lumax®), since a genetically modified and insect resistant corn variety was used. Exposure to genetically modified corn in field conditions has not been linked to detrimental effects on soil invertebrates or functions such as decomposition (Cortet et al., 2006; Hönemann et al., 2008; Tarkalson et al., 2008).

2.2. Soil physicochemical properties

Soil sampling was performed in summer 2010, three weeks after the secondary fertilization (late July), and in early fall (late September), which corresponded to the initial growth and the senescence of corn plants, respectively. Samples were taken in the four central rows of the plot using a metal core with a diameter of 45 mm diameter and length of 0.1 m. Three composite samples were taken per plot, each obtained from three soil cores.

The soil particle-distribution and texture were assessed in air-dried samples by the pipette method (Gee and Bauder, 1986). The soil organic C (SOC) content was measured by the Walkley–Black procedure (Nelson and Sommers, 1982). This method does not fully reflect C content of biochars (Manning and Lopez-Capel, 2009), but the more labile fraction (Calvelo-Pereira et al., 2011), hence potentially quantifying the most biologically relevant C fraction of biochars, potentially mineralizable by microorganisms which in turn could also affect other biological groups and soil functions.

The remaining soil properties were measured in an aqueous extract, where 25 g of fresh soil were mixed with 100 ml of deionized water and horizontally shaken at 160 rpm for 30 min. After that, soil particles were left to settle for 1 min, and the liquid phase was centrifuged for 5 min at 3600 × g. Then the supernatant was gravimetrically filtered (Whatman 1). Half of the extract was used for immediate measurement of pH and electrical conductivity (EC), while the other half was used for quantification of the ionic content (NO_2, NO_3, NH_4, PO_4, SO_4 and Cl). For practical reasons, the extract for the last analysis was stored at −20 °C just after its preparation until the day of the analysis. Simultaneously, 20 g of the same fresh soil was weighed and dried at 105 °C for 12 h for