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Effect of biochar on phosphorus sorption and clay soil aggregate stability

Helena Soinne [a,c,*], Jarkko Hovi [b], Priit Tammeorg [b], Eila Turtola [c]

[a] Helsinki University Centre for Environment (HENVI)/Department of Food and Environmental Sciences, P.O. Box 27, FIN-00014, University of Helsinki, Finland
[b] Department of Agricultural Sciences, Plant Production Sciences, P.O. Box 27, FIN-00014, University of Helsinki, Finland
[c] MTT Agrifood Research Finland, Plant Production Research, FI-31600, Jokioinen, Finland

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ABSTRACT

Soil structure is one of the key properties affecting the productivity of soils and the environmental side effects of agricultural soils. Poor surface soil structure increases the risk of soil erosion by water and eroded clay-sized particles can carry adsorbed phosphorus (P) to the surface waters, thus inducing eutrophication of receiving waterways. Management practices, e.g. reduced tillage, used to reduce erosion can lead to enrichment of P in the uppermost soil layers, which leads to elevated risk for dissolved P loss in the runoff water. In this study, we aimed to identify whether biochar (BC) could be used to reduce clay soil erosion by improving aggregate stability. Moreover, we tested whether the BC addition would change the P sorption affinity of the soil and help to reduce the loss of dissolved P. One sandy and two clayey soils were amended with BC (0, 15 and 30 t ha^{-1}) and after a 3-week incubation, a wet-sieving method was used to measure the release of colloidal particles and the stability of aggregates. The sorption of P onto soil surfaces was estimated with a Q/I (quantity/intensity) plot technique. The BC used here had a very low P sorption affinity and the BC addition did not increase the sorption of P in incubated soils. However, for the two clayey soils, the BC additions increased aggregate stability and reduced detachment of colloidal material. The BC thus induced changes in soil properties that could be beneficial for erosion control and thereby aid in reducing particulate P losses from agricultural fields.

1. Introduction

In southern Finland, about 50% of the cultivated soils are clayey soils that have received phosphorus (P) fertilization for decades. Intensive fertilization leads to high concentrations of labile P in the surface soil, increasing the risk of particulate and dissolved P losses in surface runoff (Sharpley and Withers, 1994; Turtola and Yli-halla, 1999). Disturbance of cultivated soil by tillage weakens the soil aggregate structure (Oades, 1993) and promotes erosion, which is further enhanced by lower infiltration rate and increased surface runoff. While clay particles are important for aggregate formation and a minimum of 15% clay content of the soil is needed for the abiotic development of aggregate structure (reviewed by Oades, 1993), presently in turn, the poor structure of clay soils leads to low strength under wet conditions and high risk of clay dispersion (Munkholm, 2011; Watts and Dexter, 1997). When dispersed into water, clay-sized material can travel long distances carrying pollutants and nutrients, thus inducing eutrophication of receiving waterways.

Reduced tillage has been effectively used to reduce the erosion and loss of particulate P from clayey fields (Puustinen et al., 2005; Turtola et al., 2007). However, the enrichment of P in the uppermost soil layer of the no-till soils may increase the loss of dissolved P in surface runoff (Puustinen et al., 2005; Tiessen et al., 2010). Tiessen et al. (2010) reported an increase of 12% in the annual total P export, due to the increase in dissolved P loss after converting to conservation tillage. Lehmann (2007) suggested that biochar (BC), a carbon-rich material produced by pyrolysis under anoxic conditions, can sorb phosphates, and recently results of reduced P leaching from BC-treated soils have been published. Laird et al. (2010) found that BC addition reduced P leaching after manure addition and Beck et al. (2011) found significant reduction of total P in greenroof discharge water when the soil contained 7% BC.

BC has been reported to have many positive effects on soil quality, e.g. increase in soil water retention, cation-exchange capacity (CEC) and microbial activity (Lehmann et al., 2006; Sohi et al., 2010). Similar characteristics have been attributed to the increase in natural soil organic matter (OM) content. Liang et al. (2006) showed that application of BC may result in even larger CEC than does natural OM due to its larger surface area, higher negative surface-charge and surface charge density. Natural OM plays an important role in soil structure formation and a strong positive correlation between OM and soil aggregate stability has been reported (Chaney and Swift, 1984; Heinonen, 1955; Le Bissonnais and Arrouays, 1997; Tisdall and Oades, 1982). Despite the

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* Corresponding author at: Helsinki University Centre for Environment (HENVI)/Department of Food and Environmental Sciences, P.O. Box 27, FIN-00014 University of Helsinki, Finland. Tel.: +358 9 19158324.
E-mail address: Helena.soinne@helsinki.fi (H. Soinne).

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many positive effects of BC on soil fertility, the knowledge of the effects of BC addition on soil structure and erosion is limited.

Improving the surface soil aggregate stability would aid in reducing soil erosion and mitigate the loss of particulate P from agricultural fields, while enhancing the sorption of P would reduce the risk of surface runoff loss of dissolved P. The aim of this study was to identify whether BC has the potential to a) increase soil aggregate stability and/or b) enhance sorption of phosphate–phosphorus (PO_4–P) and thus reduce the loss of particulate P and dissolved PO_4–P from cultivated soils in Finland.

2. Material and methods

2.1. Biochar

BC was produced by pyrolyzing a mixture of Norway spruce (*Picea abies* (L.) H. Karst.) and Scots pine (*Pinus sylvestris* L.) chips in a continuously pressurized carbonizer (Preseco Oy, Finland). The temperature of the carbonizer was constant at 550–600 °C during the 10–15 min pyrolysis process. After grinding, the particle size distribution (by weight) was 38% <0.25 mm, 25.9%, 0.25–1 mm, 35.9% 1–5 mm and 0.1% 5–10 mm. For the incubation study, the BC was sieved to obtain a <0.2-mm fraction that was evenly mixed with the soil volume.

The pH_{H_2O} of BC was 8.9 (1:5, v/v). The total C content was 903 g kg^{-1} (dry combustion with a VarioMax CN analyzer; Elementar Analysensysteme GmbH, Hanau, Germany). The ash content (23 g kg^{-1}) was determined by dry combustion by ramping the temperature to 500 °C in 2 h, and maintaining it for 3 h (Jones and Steyn, 1973) in a laboratory muffle furnace. The ash was dissolved with 100 mL 0.2 M HCl and boiled for 30 min. The concentrations of P and major cations determined with an inductively coupled plasma optical emission spectroscopy (Thermo-Fisher iCAP 6000; Thermo Fisher Scientific, Waltham, MA, USA) were: P: 0.2 g kg^{-1}, Ca: 4.8 g kg^{-1}, K: 2.8 g kg^{-1}, Fe: 0.4 g kg^{-1} and Mg: 0.8 g kg^{-1}. The Brunauer Emmett Teller specific surface area (BET SSA) measured using nitrogen adsorption technique with a Micromeritics Flowsorb 2300 gas adsorption analyzer was 11.8 m^2 g^{-1}.

2.2. Soil samples

The effect of BC on P sorption was tested using two fine-textured soils from Hyvinkää (silty clay loam, SiCL) and Jokioinen (Clay) (Endogleyic Stagnosol and Cutanic Vertic Luvisol, respectively, IUSS Working Group WRB, 2007), and one coarse-textured soil from Helsinki (Sand, Endogleyic Umbrisol, IUSS Working Group WRB, 2007) sampled randomly from the surface soil (Clay and Sand 0–10 cm; SiCL 0–5 cm) and mixed to form a bulked sample (Table 1). The aggregate stability tests were done only with the fine-textured soils (Clay and SiCL). The field moist samples were sieved (5 mm), homogenized and stored at +5 °C prior to the experiment.

2.3. Incubation experiment

The incubation experiment was carried out in the laboratory using plastic 300-cm^3 incubation containers. We used three BC application levels corresponding to 0, 15 and 30 t ha^{-1} mixed in the 20-cm surface layer and two moisture treatments; half of the incubation containers were kept moist and half were left to dry freely during the incubation and remoistened before the end of the incubation period. Thus, for each soil we used three BC rates with two moisture treatments replicated three times. In the containers, 150 g soil (dry matter, DM) was weighed and in the treatments with added BC it was mixed in the soil evenly. Next, the soil in the containers was compacted with 200 g cm^{-2} pressure to obtain the following bulk densities: Clay, 0.89; SiCL, 0.72; Sand, 0.96 g cm^{-3}. The soil–water content during the incubation was set to field capacity (FC; volumetric water content in pF 2 determined prior to the experiment from compacted unamended soils with a pressure-plate apparatus, corresponding to 25%, 32% and 18% for Clay, SiCL and Sand, respectively). The water content was adjusted by adding deionized water gently to the soil surface. The containers were covered with plastic film and incubated at constant temperature (+20 °C). After 4 days at constant moisture, half of the incubation containers were uncovered and left to dry freely for 16 (Clay and SiCL) or 19 (Sand) days to reach moisture contents of 10% for Clay, 15% for SiCL and 5% for Sand. During the last 3 days of the incubation, the dried soils were slowly remoistened to the FC. The remaining containers were kept at FC throughout the 20-day (Clay and SiCL) or 23-day (Sand) incubation. The moisture content in the containers during the incubation was monitored with daily weighing, and deionized water was added accordingly. Finally, the incubation containers were inverted to loosen the structure, and the pH and electrical conductivity (EC) were measured (soil:water 1:2.5, taking account of the initial moisture of the soil).

Table 1
Organic carbon (OC) content and texture of soils used in the incubation study.

Soils	Texture			OC (%)
	Sand (%)	Silt (%)	Clay (%)	
Clay	13	39	48	2.3
SiCL	3	60	38	6.7
Sand	92	6	2	2.8

2.4. Aggregate stability test

Following incubation, the aggregate stability of the amended clayey soils was tested with the wet-sieving method as follows: 4-g aggregates (dry weight, DW) were placed on a 0.25-mm sieve and left to stand in 100 ml of deionized water for 15 min. After a 15-min saturation time, the wet-sieving apparatus (Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands) was set in motion and the aggregates were dipped into the water about 95 times for 3 min. The water and detached soil material were then transferred into centrifuge tubes and the suspension was left to settle for 21 h. A 25-ml sample was pipetted from the surface of the settled suspension into a turbidimeter cuvette and the turbidity was measured by a HACH 2100N turbidimeter (Hach Co. Loveland, CO, USA). The suspension was returned to the centrifuge tube and centrifuged for 10 min (2600 *g*). The settled soil was further wet-sieved into <0.06-mm and 0.06–0.25-mm size fractions. The soil fractions were oven-dried (105 °C) and weighed to determine their masses. The mass of water-stable aggregates (WSA) was calculated as the difference between the mass originally taken for the aggregate stability analysis and the mass of the soil material detached during the wet-sieving.

2.5. Phosphorus sorption

One gram of incubated soil or soil + BC (DM) was shaken for 21 h with 50 ml of deionized water, centrifuged and filtered through a 0.2-μm Nuclepore© polycarbonate filter to determine the water-Extractable P (P_w) content. For the Q/I-plot, one gram of soil or soil + BC (DM) was mixed with 50 ml of a P solution containing 0, 0.5, 1, 1.5 and 2 mg P l^{-1}. After a 21-h equilibration, the suspensions were centrifuged and filtered through a 0.2 μm Nuclepore© polycarbonate filter. For BC, the experiments were performed with a BC-to-solution ratio of 1:100, using P solutions of 0, 1 and 2 mg P l^{-1}. The filtrate P concentrations were measured with a standard colorimetric flow injection analysis using a Lachat QuickChem 8000 (Lachat Instruments, Hach Co., Loveland, CO.). The amount of adsorbed or desorbed P was calculated from the difference in the P concentrations in the solution before (I0) and after (I) the equilibration. For each incubation container, a