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Combining phytoextraction and biochar addition improves soil biochemical properties in a soil contaminated with Cd

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HIGHLIGHTS

- Biochar increases overall enzyme activity in a soil contaminated with Cd.
- Results were enzyme specific.
- Changes in enzyme activity are not exclusively driven by alterations in soil pH.
- Synergistic effects between plant and biochar on soil biological activity are plausible.

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ABSTRACT

The main goal of phytoremediation is to improve ecosystem functioning. Soil biochemical properties are considered as effective indicators of soil quality and are sensitive to various environmental stresses, including heavy metal contamination. The biochemical response in a soil contaminated with cadmium was tested after several treatments aimed to reduce heavy metal availability including liming, biochar addition and phytoextraction using *Amaranthus tricolor* L. Two biochars were added to the soil: eucalyptus pyrolysed at 600 °C (EB) and poultry litter at 400 °C (PLB). Two liming treatments were chosen with the aim of bringing soil pH to the same values as in the treatments EB and PLB. The properties studied included soil microbial biomass C, soil respiration and the activities of invertase, β-glucosidase, β-glucosaminidase, urease and phosphomonoesterase. Both phytoremediation and biochar addition improved soil biochemical properties, although results were enzyme specific. For biochar addition these changes were partly, but not exclusively, mediated by alterations in soil pH. A careful choice of biochar must be undertaken to optimize the remediation process from the point of view of metal phytoextraction and soil biological activity.

1. Introduction

Since the beginning of the industrial revolution there has been an escalating trend in the use of heavy metals, which has resulted in increased contamination levels. Unlike organic contaminants, heavy metals are not degraded in the environment and can accumulate in soils and sediments. As a consequence, there is an upsurge in studies concerning soil heavy metal contamination (Khan et al., 2010; Vig et al., 2003), which constitutes a burden for the environment (Kabata-Pendias, 2010) and for human health (Järup et al., 1998). This includes complications (Laskowski, 1991) derived from increases in metal concentration as the element passes from lower to higher trophic levels a process known as biomagnification.

The background level of Cd in soils is less than 1 mg kg^{-1} (Adriano, 2001), however its presence in the environment has increased steadily in the last years as a consequence of man-made activities. The main anthropogenic sources of cadmium in the environment are coal combustion, municipal waste incineration, zinc, lead or copper smelter, electroplating, pigments production and nickel–cadmium batteries (World Health Organization, http://www.euro.who.int/en/home). Thus, the use of sewage sludges and phosphatic fertiliser in soils could increase their

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cadmium content (Kabata-Pendias, 2010). Cadmium is a pernicious heavy metal whose presence in the environment can result in an impact on soil ecosystem functioning. Cadmium also induces problems in human health ranging from cancer to Itai–Itai disease (Järup et al., 1998).

Starting from the 1990s there has been an increasing interest in using phytoremediation to improve soil quality in contaminated areas. Phytoremediation techniques are the most cost-efficient processes and enjoy a better public perception compared to ex situ decontamination techniques (Ali et al., 2013). Phytoextractors are plant species that can grow in areas heavily contaminated with metals and that can concentrate these metals in the harvestable parts. In the case of Cd, a plant species is believed to have the potential to phytoextract this element when its presence in the plant shoots exceeds concentrations of 100 mg Cd kg^{-1} shoot dry weight (Baker et al., 2000). The number of Cd hyperaccumulators is scarce compared to elements like Ni that can be accumulated by more than 300 plant species (Ali et al., 2013).

In the last years there has been an increasing interest on the effect of biochar on metalliferous plants, in particular in those species capable of accumulating Cd, but also other elements such as Zn, Pb and Tl (Houben et al., 2013; Fellet et al., 2014). With this aim in mind, these authors have explored the possibility to combine biochar and phytoremediation for environmental remediation, focusing on the fate of heavy metals. Those studies have demonstrated that biochar strongly immobilizes soil heavy metals, at least at the doses used in these experiments, and thus, plant uptake of the contaminant was to some extent impeded by the biochar. This resulted in an unsatisfactory recovery of heavy metal in the plant tissue.

A holistic approach to soil phytoremediation must be performed as the ultimate goals of soil remediation processes are both, to immobilize or reduce the amount of pollutant from the contaminated site, and to restore the capacity of the soil to perform its normal functions. In this sense, planting a vegetative cover can have a number of beneficial effects on soil, including reduced erosion and an improvement of soil quality and ecosystem functioning, while adding biochar to soil can result in an enhancement in carbon sequestration (Lehmann et al., 2006) and soil biological properties (Paz-Ferreiro et al., 2012) or in a reduction in soil erosion (Jien and Wang, 2013). These co-benefits of biochar application have not been evaluated in previous studies dealing with the interaction between biochar and metalliferous plants.

Indicators of soil quality that can properly assess the efficiency of a phytoremediation process must be chosen accordingly. There is a considerable and ever-increasing bibliography regarding the use of soil enzymes as indicators of soil quality due to their rapid response after land use changes or alterations in soil management (Paz-Ferreiro et al., 2010, 2011). In fact, several mathematical expressions using soil enzymes as indicators of soil quality, either by themselves or combined with other biological, physical or chemical soil properties have been proposed in the last years (Paz-Ferreiro and Fu, in press). In spite of this, there are few studies that tried to assess the effect of phytoremediation (Epelde et al., 2009; Moreno-Jiménez et al., 2012) or biochar (Paz-Ferreiro et al., 2012; Wu et al., 2013) separately on soil enzyme activities.

In general, heavy metals have a negative impact on soil biochemical properties and, in particular, towards soil enzymes (Hinojosa et al., 2004; Khan et al., 2010), although this effect can be different from an enzyme to another and depend also on the pollutant (Shen et al., 2005; Khan et al., 2010). Most soil enzymes have negative relationships with increasing amount of extractable heavy metal (Hinojosa et al., 2004) and it is a well-known fact that soil restoration can improve the biochemical activity of a contaminated area (Hinojosa et al., 2004).

Thus, the aim of our work is to study the soil biochemical response after the use of biochar and phytoextraction for remediation purposes. In addition we employed liming treatments to assess the contribution of pH changes to alterations in soil biological properties mediated by biochar, as due to the frequently reported proximity between organisms and biochar surfaces, as reviewed by Lehmann et al. (2011), biochar pH could have a key influence on total microbial abundance. We hypothesized that, in spite of the effectiveness of phytoextraction being diminished in this soil by biochar use (Lu et al., 2014), soil biochemical quality could benefit from a combination of phytoextraction and biochar addition. We also hypothesized that biochars prepared from different feedstocks and at different temperatures would affect differently enzyme activity patterns.

2. Materials and methods

The soil and experimental design used in this study have been described previously (Lu et al., 2014). Basically, soil was collected from the surface layer (0–20 cm) of a cropland area. Sampling took place in 20–25 points over a 0.5 hectare area, totaling an amount of 50 kg of soil. The sampling area was located near a waste landfill site in the suburb of Guangzhou, China. Guangzhou is located in the subtropical humid area having an average annual temperature of 12.7 °C and annual average precipitation of 1700 mm. According to FAO, the soil is classified as a Fimic Anthrosol.

Part of the soil was air dried and sieved to 2 mm to conduct general analyses, while the rest was moist sieved to 10 mm to conduct the experiment. Before the starting of the pot experiment, the soil had a total organic carbon content of 1.98%, total nitrogen content of 0.142%, pH of 6.00, total phosphorus of 687 mg kg^{-1}, available phosphorus of 126 mg kg^{-1}. Soil had a sandy-loam texture, with 17% clay, 7% of silt and 76% of sand. Its total Cd content was 6.1 mg kg^{-1}, a figure more than 20 times higher than the Chinese Soil Environmental Quality Standard Guide value of 0.3 mg kg^{-1} (China GB 15618-1995, 1995). At this level of Cd contamination Anthrosols in South China show a reduction of soil microbial properties, including soil enzymes (unpublished data from the authors).

2.1. Preparation and characterisation of biochar

The two biochars used in the present experiment were obtained from poultry litter and eucalyptus as feedstocks (PLB and EB, respectively). The preparation of these materials, the methods to characterise the biochars and the reasons to select these materials are described in more detail in Lu et al. (2014), while the main properties of the biochars are shown in Table 1.

The concentration of total polycyclic aromatic hydrocarbons (PAH) was determined as in Tammeorg et al. (in press). Briefly, Soxhlet extractions (0.5 g of biochar, 90 mL of toluene, 6 h, 160 °C) were spiked with 1,1-binaphthyl as an internal standard before extraction. The toluene was reduced to 1 mL and the content of 18 PAHs (naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[j] fluoranthene benzo[k]fluoranthene benzo[a]pyrene, indeno[1,2, 3-cd]pyrene, dibenz[a,h]anthracene, benzo[g,h,i]perylene, benzo[e]pyrene) were determined by gas chromatography mass spectroscopy analysis.

For heavy metal analysis, samples (0.2 g) were digested with 6 mL HNO_3 and 2 mL H_2O_2 using a microwave closed system (Multiwave3000, Anton Paar, Austria). Heavy metal concentrations were analysed using an ICP-MS spectrometer (Agilent 7700x, USA).