Crop response to soils amended with biochar: expected benefits and unintended risks

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Abstract

Biochar (BC) from biomass waste pyrolysis has been widely studied due to its ability to increase carbon sequestration, reduce greenhouse gas emissions, and enhance both crop growth and soil quality. This review summarises the current knowledge of BC production, characterisation, and types, with a focus on its positive effects on crop yield and soil properties vs the unintended risks associated with these effects. Biochar-amended soils enhance crop growth and yield via several mechanisms: expanded plant nutrient and water availability through increased use efficiencies, improved soil quality, and suppression of soil and plant diseases. Yield response to BC has been shown to be more evident in acidic and sandy soils than in alkaline and fine-textured soils. Biochar composition and properties vary considerably with feedstock and pyrolysis conditions so much that its concentrations of toxic compounds and heavy metals can negatively impact crop and soil health. Consequently, more small-scale and greenhouse-sited studies are in process to investigate the role of BC/sol/crop types on crop growth, and the mechanisms by which they influence crop yield. Similarly, a need exists for long-term, field-scale studies on the effects (beneficial and harmful) of BC amendment on soil health and crop yields, so that production guidelines and quality standards may be developed for BCs derived from a range of feedstocks.

Introduction

In the agricultural sector, the term biochar (BC) is acknowledged as a tool for soil management and carbon (C) sequestration (Barrow, 2012; Jeffery et al., 2015b; Smith, 2016). It is a solid material, rich in recalcitrant C, derived from anaerobic heating of biomass such as wood, manure, sludge, or crop residue (Lehmann and Joseph, 2015). The use of BC for soil C sequestration followed the discovery of so-called terra preta, the charcoal-rich and highly fertile soil of the central Amazon basin (Barrow, 2012). Biochar, a secondary product of pyrolysis after syngas and bio-oil production for bioenergy (Srivasan et al., 2015; Jeffery et al., 2015b), is touted for its diverse and positive agricultural roles. Its C sequestration (Lehmann and Joseph, 2015) and greenhouse gas (GHG) emission reduction (Cayuela et al., 2014; Ameloot et al., 2013a, 2013b; Smith, 2016) help mitigate climate change and potential adverse impacts to ecosystems from agriculture (Barrow, 2012; Liu et al., 2014). Additionally, it (Jensen 2013a, 2013b) represents an alternative means by which to contain water and atmospheric pollution ascribed to large volumes of crop and animal wastes (Jeffery et al., 2015b). As a soil amendment (Lehmann et al., 2011, 2015; Windeatt et al., 2014), BC has received increased interest due to its role in enhancing nutrient- and water-use efficiencies (Van Zwieten et al., 2010; Singh et al., 2010; Barrow, 2012). Finally, it has been found to immobilise and remove soil and water contaminants (Novak et al., 2016). Keeping in view its multifaceted benefits, BC emerges as a social, economic, and environmental friendly product (BRC, 2014).

The positive effects of amending the soil with BC include increased crop productivity and soil fertility (Liu et al., 2013, 2014; Windeatt et al., 2014; Biederman and Harpole, 2013; Jeffery et al., 2011, 2015b) mainly through improved quality, C storage, infiltration, and water holding capacity of soil (Mukherjee and Lal, 2013; Mukherjee et al., 2014). Compared to other amendments, BC is thought to be C-negative because it is derived from atmospheric CO₂ captured by plants, which is then diverted to the soil in a very stable form where it can remain for several years (Lehmann and Joseph, 2015; Smith, 2016). Authors have also reported that applying BC to agricultural soils can lead to resistance against several soil- and air-borne plant diseases (such as potato rot, tomato seedling damping-off, pepper and strawberry fungal diseases, and carrot root-lesion nematode), through stimulation of several general defence pathways and promotion of defence-related gene expression (Elad et al., 2011; Harel et al., 2012; George et al., 2016).

Despite its many benefits, BC also poses risks, associated with the properties of the original feedstock and thermal process conditions (Deenik et al., 2010; Spokas et al., 2011; Freddo et al., 2012; Barrow, 2012; Quilliam et al., 2013b; Buttner et al., 2015; Camps-Arbestain et al., 2015; Domene et al., 2015; Genesio et al., 2016). These can range from heavy metal soil accumulation to toxic compound release, and to other negative effects on soil biota and...
human health. This review paper analyses the current knowledge of BC production and characterisation not only to emphasise its role in enhancing crop yield, but also to unleash its potential risks. This paper has been sectioned as follows: BC properties, impacts on soil quality, and selected crops productivity as a function of BC properties, application rates, post-application elapsed time, and external environmental conditions. Furthermore, we examine the undesirable soil and crop effects that may occur following BC application to agricultural soils.

Biochar production and properties

Pyrolysis processing conditions

Biochar can be produced from a variety of feedstocks (Cantrell et al., 2012; Jeffery et al., 2015b; Ronsse et al., 2013). The conditions for its optimal production with improved nutrient properties differ with the original feedstock (manure, sewage sludge, biosolid, crop residue, or wood biomass), thermal process used (pyrolysis, gasification, or hydrothermal carbonisation (HTC)), and pyrolytic conditions (slow/fast or low/high temperature) (Figure 1; Spokas et al., 2012; Bridgewater, 2012; Mukome et al., 2013; Li et al., 2016). Biochar production can be optimised at pyrolytic temperatures (PT) between 300 and 800°C. Gasification of biomass occurs at temperatures above 750°C and with limited oxygen (<10%) supply to facilitate burning of the biomass. Generally, BC yield falls with rising PTs, as weak-bound C gets volatilised at high temperature (Table 1; Cantrell et al., 2012). Also, slow pyrolysis at low PTs (<500°C) generally produces higher BC yields than does fast pyrolysis at high PTs, but the resulting BCs contain less aromatic C than those produced at high PTs (Ronsse et al., 2013).

Conversely, fast and intermediate pyrolysis at greater PTs (>500°C), with short residence time of 1-30 seconds, produce more liquid (bio-oil) products (50−75%) and higher fixed-C-containing BC whereas more solid (char) and gaseous (syngas) product (up to 35% each) is obtained through slow pyrolysis at low PTs with long residence time (>30 minutes) (Figure 1; Bridgewater, 2012; Enders et al., 2012). Nonetheless, it is feedstock choice and market conditions that often drive optimisation of pyrolytic conditions.

Hydrochar, an HTC product, contrasts with BC in that it is produced from feedstock pyrolysis under subcritical conditions (liquid water), called wet pyrolysis (Libra et al., 2011; Subedi et al., 2015). The process normally operates at pressures and temperatures of 15-25 MPa and 180-250°C, respectively (Kammann et al., 2012). The C-content of hydrochar is lower relative to BC as some biomass-C leaches as dissolved organic carbon (DOC) during the reaction with water, although both the C and N gaseous losses are almost negligible (Appendix Table 1; Kammann et al., 2012; Subedi et al., 2015). It is often reported that HTC operating conditions never attain spontaneous carbonisation temperatures, and thus, remain endothermic. However, we know even less of hydrochar production, characterisation, and application than we do BC (Libra et al., 2011).

Biochar properties as a function of processing conditions

Appendix Table 1 shows that BC properties are governed by feedstock type and composition (Singh et al., 2010; Cantrell et al., 2012; Camps-Arbestain et al., 2015; Subedi et al., 2015; 2016a, 2016b), thermal process (pyrolysis, gasification, HTC) (Libra et al., 2011; Lynch et al., 2013), pyrolytic conditions (slow/fast, high/low PT) (Bridgewater, 2012), and operating conditions (temperature, heating rate, residence time) (Lee et al., 2013). Of most significance are its properties of surface area, pH, and nutrient composition.

Table 1. Changes in biochars properties as influenced by pyrolysis conditions.

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<th>Physico-chemical parameters</th>
<th>Effect/changes due to PT</th>
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<td>Spokas (2010), Budai et al. (2014)</td>
<td>BC yield</td>
<td>Decreases with increase in PT</td>
</tr>
<tr>
<td>Lehman (2007), Ippolito et al. (2015)</td>
<td>C recovery</td>
<td>Decreases with increase in PT due to volatilisation of C at high PT</td>
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<td>Cantrell et al. (2012), Subedi et al. (2016b)</td>
<td>N recovery</td>
<td>Decreases with increase in PT due to volatilisation of N at high PT</td>
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<td>Singh et al. (2010), Cantrell et al. (2012), Wang et al. (2012b)</td>
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<td>Increases with increase in PT due to increased recovery of P in the ash fraction</td>
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<tr>
<td>Cantrell et al. (2012), Subedi et al. (2016b)</td>
<td>S recovery</td>
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<td>Singh et al. (2010), Budai et al. (2014)</td>
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<td>Lehman (2007), Budai et al. (2014)</td>
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<td>Singh et al. (2010), Mukome et al. (2013), Subedi et al. (2016b)</td>
<td>Surface acidity</td>
<td>Decreases with increase in PT due to loss of acidic functional groups at high PT</td>
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<td>Spokas et al. (2011), Subedi et al. (2016b)</td>
<td>VM</td>
<td>Decreases with increase in PT</td>
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<td>Lehman (2007), Budai et al. (2014), Mukome et al. (2013)</td>
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<td>Increases up to 500°C followed by decrease (&gt;500°C) due to loss of acidic functional groups</td>
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<td>Lee et al. (2010), Fuertes et al. (2010), Chia et al. (2015)</td>
<td>Porosity</td>
<td>Increases up to 600°C (anti-clogging of pore space) followed by decrease (&gt;600°C) due to collapse of pore and surface structures</td>
</tr>
<tr>
<td>Lehman (2007), Lee et al. (2010), Budai et al. (2014)</td>
<td>SA</td>
<td>Increases up to 600°C followed by decrease (&gt;600°C) due to collapse of pore structure</td>
</tr>
<tr>
<td>Singh et al. (2010), Cantrell et al. (2012), Subedi et al. (2016b)</td>
<td>Cations (Ca, Mg, K, Na)</td>
<td>Increases with increase in PT due to increased recovery of cations in ash fraction</td>
</tr>
<tr>
<td>Camps-Arbestain et al. (2015), Domene et al. (2015)</td>
<td>Heavy metals</td>
<td>Increases with increase in PT due to increase in ash content</td>
</tr>
</tbody>
</table>

PT, pyrolysis temperature; BC, biochar; C, carbon; N, nitrogen; P, phosphorus; S, sulphur; VM, volatile matter; CEC, cation exchange capacity; SA, surface area; Ca, calcium; Mg, magnesium; K, potassium; Na, sodium; Fe, iron.
Biochar is highly porous due to retention of its feedstock cell wall structure (Chia et al., 2015). Generally, BCs of larger pore diameters have greater surface areas and lower bulk densities (Mukherjee et al., 2011; Jeong et al., 2016). Biochar surface area enlarges as PTs rise because the surface organic molecules that obscure its pores at low PTs (< 500°C) are volatised at high temperatures (Table 1; Lehmann 2007; Fuertes et al., 2010).

Biochar pH generally turns more alkaline at high PTs because high heat increases both its ash content and loss of acidic functional groups (Table 1 and Appendix Table 1; Singh et al., 2010; Ippolito et al., 2015). Functional groups (-OH, -COOH groups) that primarily determine BC reactivity and raise soil CEC, collapse at high PTs (> 500°C) (Table 1; Cheng and Lehmann, 2009; Singh et al., 2010; Subedi et al., 2016a, 2016b).

The nutrient composition of BC is derived from feedstock type and composition and processing conditions (Table 1 and Appendix Table 1; Camps-Arbestain et al., 2015). On the other hand, nutrient retention and availability relate to BC adsorption capacity (Singh et al., 2010; Ippolito et al., 2015; Zornoza et al., 2016). Overall, manure- and sludge-based BCs are richer in plant available nutrients as compared to grass- and wood-based BCs, which is mostly a reflection of their original feedstock properties, although C, N, and S losses at high PTs are unavoidable (Cantrell et al., 2012; Jensen, 2013b; Subedi et al., 2016a, 2016b). Other nutrients (P, K, Ca, Mg, and Na) found mainly in the ash fraction, are plant-available depending on the soil-BC matrix, pH, and the presence of chelating substances that regulate their mobilisation (Cui et al., 2011; Wang et al., 2015). Nitrogen content and availability correlate highly to PT, rate of heating, and feedstock type and composition (Wang et al., 2012a; Clough et al., 2013; Subedi et al., 2016b). Differential response to PTs showed that at low PT (<400°C) BCs favour greater recovery of C and other nutrients (both macro and micro) compared with high PT (>400°C) BCs because these nutrients are increasingly lost at higher PTs. In addition, low PT BCs have faster and greater soil activity, and thus contribute more to soil fertility (Joseph et al., 2010).

### Biochar effects on soil properties

It is recognised that BC resists microbial attack better than almost any other soil amendment (Lehmann et al., 2011). Despite containing a high fraction of very stable compounds, the unique physical and chemical properties of BC can improve soil (Barrow et al., 2012; Singh et al., 2010; Mukome et al., 2013). The specific soil-BC combination affects amended soil properties (Cantrell et al., 2012; Camps-Arbestain et al., 2015). In fact, several authors have shown that BC amendment can positively modify the physical, chemical, and biological properties of soil (Barrow, 2012; Mukherjee and Lal, 2013; Subedi et al., 2016b; Schmalenberger and Fox, 2016).

### Soil physical properties

As a consequence of improved soil physical properties (structure, surface area, porosity, bulk density, and water holding capacity), plant water availability, nutrient retention capacity, root penetration, and aeration do increase (Chia et al., 2015; Mukherjee and Lal, 2013). Sandy soils amended with BC have higher water holding capacities than do loamy and clay soils, while increased soil aeration is mainly observed in fine-textured soils (Mukherjee et al., 2014; Jeffery et al., 2011; Liu et al., 2013). As a result of BC addition, the great surface areas of amended soils can favour microbial communities and overall soil sorption capacities; addi-
tionally, the high internal surface area of BC improves water retention directly and soil structure indirectly (Mukherjee and Lal, 2013).

**Nutrient availability**

Nutrient availability in BC-amended soils is often not only associated with the physico-chemical properties of the char (Clough and Condon, 2010; Ippolito et al., 2012, 2015; Mukome et al., 2013), but also directly linked to nutrient availability. Mechanisms responsible for increasing plant nutrient availability are soil pH raise (in acidic soils), nutrient retention (due to increase in CEC and surface area) or directly release of nutrients from the BC surfaces (DeLuca et al., 2015; Clough et al., 2013; Subedi et al., 2016b). Liard et al. (2010) has demonstrated enhanced nutrient retention from soils amended with BC. In fact, BC C has been shown to oxidise in soil and raise soil CEC after just a few years of application (Cheng and Lehmann, 2009; Singh et al., 2010), even though it remains unclear as to how the CEC of BC changes as it disintegrates during tilling and weathering.

We do know that biotic and abiotic factors greatly influence nutrient transformation and mobilisation in a BC-soil matrix (Clough et al., 2013; DeLuca et al., 2015; Schmalenberger and Fox, 2016). Commonly available commercial BCs, derived from wood biomass and crop residues, are poor in nutrients, but can positively affect nutrient availability and crop growth only if enriched with nutrients pre-application or if applied in combination with fertilisers (Deenik et al., 2010; Kammann et al., 2015; Yao et al., 2015). Manure and biosolid-derived BCs, which are used less often than wood BCs, can function as both amendments and biointegrators for their nutrient-rich original feedstocks (Singh et al., 2010; Hossain et al., 2015; Cantrell et al., 2012; Subedi et al., 2016a, 2016b).

**Soil pH**

Having alkaline nature, BC acts as a timing agent in acidic soils (Alburquerque et al., 2013; Ameloot et al., 2013a; Subedi et al., 2015, 2016a, 2016b). The effect can be heightened if poultry litter-derived BC, which has high concentrations of carbonate, is used as an amendment (Chan et al., 2008; Van Zweiten et al., 2010). Manure- and sludge-based BCs produce the same strong soil-liming effect, due to high concentrations of alkali metals and exchangeable basic cations (Ca, Mg, K and Na) present in their ash fractions, relative to wood biomass-derived BCs (Appendix Table 1; Singh et al., 2010; Enders et al., 2012; Domené et al., 2015; Srinivasan et al., 2015).

Furthermore, BC is also thought to have potential in improving reforestation in saline-sodic soil possibly through minimisation of salt toxicity in such soils (Drake et al., 2016). This means that BC has a potential for the reclamation of saline-sodic soils. On the contrary, some authors have reported no effect on soil pH following BC addition, particularly when produced at low PT (<350°C) and when amending either alkaline or saline soils (Novak et al., 2009; De La Rosa et al., 2014; Olmo et al., 2014). This may be reflective of the strong influence of PT on BC pH (Janus et al., 2015), and also of the soil CaCO₃ buffering effect (Olmo et al., 2014).

**Soil microorganisms**

Biochars are fully sterilised during pyrolysis, thus any direct contribution to microbial populations is their elimination (Lehmann et al., 2011; Thies et al., 2015). Instead, the high porosity of BC may indirectly increase soil microbial biomass and basal activity (Lehmann et al., 2011) by providing a protected and aerated habitat for microbial growth (Fox et al., 2014; Schmalenberger and Fox, 2016). In other instances, BC directly modifies soil microbial community structure by providing nutrients (Subedi et al., 2015; Schmalenberger and Fox, 2016). The large internal surface area of BC expands the organic and inorganic compound adsorption capability of soil, such that the supply of mineral nutrients and energy to microbes is increased (Lehmann et al., 2011; Gul et al., 2015). While it has long been reported that soil biodiversity and soil organic matter are positively correlated, the role of BC in the interaction requires further investigation (Ameloot et al., 2013b; Kuppusamy et al., 2016).

**Biochar stability in soils**

As evidenced by Amazonian dark earth soils, it is fairly clear that BC is more stable than any other soil amendment due to its aromatic structure (some compounds have residence time >1000s of years) (Lehmann and Joseph, 2015). However, biotic and abiotic degradation after soil application occurs and its intensity is driven by its C:N ratio and labile C content. Besides as CO₂ via oxidation, BC can also be lost with erosion, surface run-off, and leaching of percolating water as DOC due to excessive rainfall, as reported for a Colombian savanna Oxisol (Major et al., 2010). Ultimately, BC ageing and movement through the soil profile are fertile grounds for research of its long-term effects on physical, chemical, and microbial soil properties (Singh et al., 2010; Clough and Condon, 2010; Ippolito et al., 2012, 2015; Kuppusamy et al., 2016).

**Biochar effects on crop productivity**

Soil amendment with BC is expected to increase crop productivity by enhancing the supply of nutrients and by fostering the activity of soil microorganisms responsible for mobilising soil nutrients and making them more available to crops (Lehmann et al., 2011, 2015; Liu et al., 2013; Camps-Arbestain et al., 2014; Schmalenberger and Fox, 2016) and promoting root expansion. Crop growth promotion in BC-amended soils is mainly linked to increased nutrient use efficiency, reduced nutrient leaching, and improved soil physical (Mukherjee and Lal, 2013; Mukherjee et al., 2014), chemical (De La Rosa et al., 2014; Subedi et al., 2016b), and microbial properties (Nielsen et al., 2014; Gul et al., 2015).

Many recent studies have reported increased crop yield (Subedi et al., 2016a; Vaccari et al., 2011; Usman et al., 2016; Kammann et al., 2012; Baronti et al., 2010; Uzoma et al., 2011; Houben et al., 2013; Genesio et al., 2015; Cornißen et al., 2013; De La Rosa et al., 2014; Fox et al., 2014; Gregory et al., 2014; Lin et al., 2015; Schmidt et al., 2015; Butman et al., 2015; Laghari et al., 2015; Mandal et al., 2016). Others have reported no yield effect (Cornißen et al., 2013; Uzoma et al., 2011; Nelissen et al., 2015; Subedi et al., 2016a, 2016b; Nielsen et al., 2014; Tammeorg et al., 2014; Suddick and West, 2013; Schmidt et al., 2014; Bass et al., 2016). Conversely, few studies have instead described reduced crop yields (Deenik et al., 2010; Baronti et al., 2010; Marks et al., 2014; Nelissen et al., 2014; Bass et al., 2016; Butman et al., 2015; Laghari et al., 2015). Therefore, interest has emerged to study BC effects that negatively alter crop growth (Jeffery et al., 2011, 2015; Spokas et al., 2012; Biederman and Harpole, 2013; Wang et al., 2016, Olmo et al., 2016). Factors responsible for yield response to BC are specific BC/soil/crop/fertiliser combination, application rate, elapsed
incorporation time, experiment type, and environmental conditions (Jeffery et al., 2011, 2015a; Biederman and Harpole, 2013; Liu et al., 2013; Lychuk et al., 2015). The qualitative data presented in Table 2 display the results from several recent studies conducted in open fields and greenhouses on the effects of BC on crop productivity.

**Interaction with crop type**

Yield response varies with crop type, and in general, interaction between crop and BC type cannot be underestimated based on the following study results (Jeffery et al., 2011, 2015a; Biederman and Harpole, 2013; Liu et al., 2013). Lin et al. (2015) observed yield increases of 11% in soybean grain yield and of 28% in wheat grain yield following maize stalk BC application to a coastal saline soil. In an Italian vineyard soil, Baronti et al. (2014) demonstrated an increase in leaf water potential of 24-37%, which reduced water stress in the grape crop and lead to an improved water use efficiency. Genesio et al. (2015) reported an even greater grape yield increase (66%) in the same field after applying BC from orchard prunings. On the other hand, Schmidt et al. (2014) reported neither a grape yield nor quality effect after wood BC was applied to Swiss vineyard soils during a four-year field trial. A two-year field study in Belgian sandy soil with sub-acidic pH by Nelissen et al. (2015) also found no effect on either spring barley or winter rye crop yields after wood BC application (22 t ha⁻¹). Jones et al. (2012) conducted an open field experiment in the UK and found no effect on maize yield in the first year of application on a near-neutral pH sandy clay-loam soil, but observed a 30% increase in hay grass yield in the second and third years.

In an eight-week greenhouse trial of a mixed cropping system (red clover, fescue, and plantain) conducted on BC-amended acidic, podzol soil, Oram et al. (2014) discovered a 28-50% increase in biomass yield that fell to between 8 and 30% in a monoculture system under the same conditions. These results highlighted the effectiveness of BC to enhance yield in mixed-culture as opposed to monoculture.

Biochar also has demonstrated specific effects on legume N fixation capacity. Rondon et al. (2007), in a pot experiment, found common bean yield increased by 46% and biological N fixation increased by 44% when eucalyptus BC was added (9% w/w) to Columbian oxisol which could be ascribed to several factors, such as improved B and Mo availability, lower N availability, increased K, Ca, and P availability, decreased exchangeable Al, and higher pH. A further greenhouse study of Güereña et al. (2015) confirmed this hypothesis, based on results of an average increase of 162% in common bean biomass yield, and an 18-fold increase in fixed N from the atmosphere following the application of BCs from a variety of feedstocks (sugarcane bagasse, wood, tea pruning, maize stover/cobs, and rice hull). Quilliam et al. (2013a), in a three-year field trial in a British sandy clay-loam soil with near neutral pH, found that mixed-wood BC applied at 25-50 t ha⁻¹ significantly increased nitorgenase activity (with no effect on nodulation) in sweet clover, if compared with non-amended soil.

**Interaction with soil type**

Effects of BC on crop yield also depend on soil type. In a trial in Zambian, Cornelissen et al. (2013) found maize yields grew considerably (233-322%) after application of soft wood- and maize cob-derived BC (4 t ha⁻¹) on an acidic sandy soil, moderately (30-42%) on a sandy clay-loam acidic soil, and without significant effect on a clay-loam neutral, an acidic sandy loam and a silty-clay soil. The authors ascribed the large maize yield in the acidic sandy soil to increased soil CEC, base saturation, plant available water, and water use efficiency. Subedi et al. (2016a) observed up to a 50% increase in ryegrass yield in sub-acid silt-loam soil and a 44% increase in calcareous sandy soil. In an Australian calcareous soil amended with pine and poplar wood BCs, Marks et al. (2014) reported negative effects on lettuce and ryegrass yields. The authors reported this as restricted nutrient availability for the crops limited by the increased VM content of BC that increased competition with the microorganisms plus phosphate precipitation to non-available forms due to biochar chemistry.

Interactions with soil characteristics are further complicated by the effects of BC production technology options. In a greenhouse trial in Thailand, Butnan et al. (2015) found that in the first growth cycle maize yield decreased in acidic loamy-sand amended with a Eucalyptus wood-based BC that underwent flash carbonisation at high PTs (800°C), while no effect was shown with the same BC in sub-acidic silt-clay-loam soils and on both soils using a BC kilned at low PT (350°C). However, results changed in the second crop cycle, as yields arose as much as 600% in the silty-clay-loam soil and 250% in the loamy-sand soil amended with low PT BC. The authors speculated that yield decreased in the soil amended with the flash-carbonised-char came from the deleterious effects on plant growth of polycyclic aromatic hydrocarbons (PAHs) (amount not quantified), and to the antagonistic effects of high-K content on Ca²⁺ and Mg²⁺. They attributed the second crop cycle increased yield to BC ageing in the soil that increased soil CEC, nutrient availability, water-holding capacity, liming, and reduced Al and Mn toxicity (Singh et al., 2010; DeLuca et al., 2015; Butnan et al., 2015).

**Interaction with organic and inorganic fertilisers**

As most commercial BCs derived from wood biomasses, orchard pruning residues, green wastes, with the exception of manure and sludge biomasses, are poor in nutrient compositions, studies have shown that BC affect positively on crop growth when applied in combination with fertilisers i.e. organic as well as inorganic (Steiner et al., 2007; Deenik et al., 2010; Subedi et al., 2015). This is probably due to the positive interaction between BC and applied fertiliser that improved the availability of nutrients associated with enhanced plant uptake and reduced losses of these nutrients. Schmidt et al. (2015) reported an 85% increase in pumpkin crop yield versus the control following soil application of *Eupatorium* weed-derived BC. The yield rose to 300% when cattle urine was added to this BC before soil application. Similar yield increases have been reported by Baronti et al. (2010) in maize, Kammann et al. (2015) in *Chenopodium*, and Alburquerque et al. (2013) in wheat when BC was combined with either organic residues/compot or mineral fertiliser, and indicate that wood BC may raise nutrient use efficiency when added to organic/inorganic fertiliser/crop residues.

**Possible causes for negative crop response**

A normal BC application rate – in the range 5-20 t ha⁻¹, similar to other amendments such as compost – under normal conditions can positively affect crop yield, although correctly matching a BC, with its given set of properties, to a specific soil type is a very important agronomic decision.

Excessive application rates (>50 t ha⁻¹) may negatively affect crop response, but it is difficult to define an exact threshold above which negative effects appear, as they are a result of BC characteristics besides practical constraints due to product availability and...
Table 2. Effect of selected biochars on productivity of different crops.

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<th>Location</th>
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<th>Process condition*</th>
<th>Application rate</th>
<th>Type of experiment</th>
<th>Duration</th>
<th>Crop grown</th>
<th>Soil type</th>
<th>Fertiliser coaddition</th>
<th>Effects on crop yield</th>
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<tr>
<td>Vaccari et al. (2011)</td>
<td>Italy</td>
<td>Mixed hardwood</td>
<td>PY, 500</td>
<td>30 and 60 t ha⁻¹</td>
<td>F</td>
<td>2 crop cycles</td>
<td>Durum wheat</td>
<td>AC silt-loam</td>
<td>IN</td>
<td>+30% (grain)</td>
</tr>
<tr>
<td>Subedi et al. (2010a)</td>
<td>Italy</td>
<td>Poultry litter, swine manure</td>
<td>PY, 400 and 600</td>
<td>20 g kg⁻¹</td>
<td>P</td>
<td>20 weeks</td>
<td>Italian rye grass</td>
<td>AC silt-loam and AK sandy</td>
<td>0</td>
<td>+ (18-50)% (biomass)</td>
</tr>
<tr>
<td>Subedi et al. (2010b)</td>
<td>Italy</td>
<td>Wood chips</td>
<td>GA, 1000</td>
<td>20 g kg⁻¹</td>
<td>P</td>
<td>20 weeks</td>
<td>Italian rye grass</td>
<td>AC silt-loam and AK sandy</td>
<td>0</td>
<td>Neutral</td>
</tr>
<tr>
<td>Laghari et al. (2015)</td>
<td>China</td>
<td>Pine sawdust</td>
<td>Fast PY, 400</td>
<td>22 t ha⁻¹</td>
<td>P</td>
<td>8 weeks</td>
<td>Sorghum</td>
<td>AK, sandy (desert soil)</td>
<td>IN</td>
<td>+ (18-22)% (biomass)</td>
</tr>
<tr>
<td>Laghari et al. (2015)</td>
<td>China</td>
<td>Wheat straw, enriched with nutrients</td>
<td>PY, 450</td>
<td>670 kg ha⁻¹</td>
<td>F</td>
<td>15 weeks</td>
<td>Green pepper</td>
<td>CLAY, sandy</td>
<td>IN</td>
<td>+ (16-18)% (fruit)</td>
</tr>
<tr>
<td>Cornelissen et al. (2010)</td>
<td>Belgium</td>
<td>Mixed hardwood</td>
<td>PY, 500</td>
<td>10 t ha⁻¹</td>
<td>F</td>
<td>1 crop cycle</td>
<td>Durum wheat, maize</td>
<td>AK, clay-loam and AC, silty-loam</td>
<td>IN, ORG</td>
<td>+ (10-30)% (fruit)</td>
</tr>
<tr>
<td>Baronti et al. (2010)</td>
<td>Italy</td>
<td>Mixed hardwood</td>
<td>PY, 500</td>
<td>30 and 60 t ha⁻¹</td>
<td>P</td>
<td>8.5 weeks</td>
<td>Perennial ryegrass</td>
<td>Sandy-loam, sub-AC</td>
<td>0</td>
<td>+ (29-30)% (biomass)</td>
</tr>
<tr>
<td>Baronti et al. (2010)</td>
<td>Italy</td>
<td>Mixed hardwood</td>
<td>PY, 500</td>
<td>100 and 120 t ha⁻¹</td>
<td>P</td>
<td>8.5 weeks</td>
<td>Perennial Ryegrass</td>
<td>Sandy-loam, sub-AC</td>
<td>0</td>
<td>+ (18-20)% (biomass)</td>
</tr>
<tr>
<td>Schmidt et al. (2015)</td>
<td>Nepal</td>
<td>Eupatorium adenophorum</td>
<td>PY, 689-700</td>
<td>0.75 t ha⁻¹</td>
<td>F</td>
<td>17 weeks</td>
<td>Pumpkin</td>
<td>AC, silt-loam</td>
<td>ORG</td>
<td>+ (85-100)% (fruit)</td>
</tr>
<tr>
<td>Cornelissen et al. (2015)</td>
<td>Belgium</td>
<td>Mixed hardwood</td>
<td>PY, 500</td>
<td>10 t ha⁻¹</td>
<td>F</td>
<td>24 months</td>
<td>Spring barley, winter rye</td>
<td>Sub-AC, sandy loam</td>
<td>IN</td>
<td>Neutral</td>
</tr>
<tr>
<td>Nelissen et al. (2014)</td>
<td>Belgium</td>
<td>Mixed hardwood</td>
<td>PY, 450, 550 and 650</td>
<td>10 g kg⁻¹</td>
<td>P</td>
<td>5 weeks</td>
<td>Radish, spring barley</td>
<td>AC and N, sandy loam</td>
<td>IN, 0</td>
<td>+ (11-48)% (radish), (3-35)% (barley)</td>
</tr>
<tr>
<td>Nelissen et al. (2014)</td>
<td>Belgium</td>
<td>Mixed hardwood</td>
<td>PY, 450, 550 and 650</td>
<td>10 g kg⁻¹</td>
<td>P</td>
<td>5 weeks</td>
<td>Radish, spring barley</td>
<td>AC and N, sandy loam</td>
<td>IN, 0</td>
<td>+ (11-50)% (barley)</td>
</tr>
<tr>
<td>Hossain et al. (2015)</td>
<td>USA</td>
<td>Wastewater sludge</td>
<td>PY, 550</td>
<td>16 t ha⁻¹</td>
<td>GH</td>
<td>16 weeks</td>
<td>Cherry tomato</td>
<td>AC, Chromosol</td>
<td>0</td>
<td>+44% (fruit)</td>
</tr>
<tr>
<td>Rogovska et al. (2014)</td>
<td>Belgium</td>
<td>Hardwood</td>
<td>GA, 500-575</td>
<td>0.94 t ha⁻¹</td>
<td>F</td>
<td>1 crop cycle</td>
<td>Maize</td>
<td>AC, loam</td>
<td>0</td>
<td>+ (11-55)% (graing)</td>
</tr>
<tr>
<td>Edward et al. (2013)</td>
<td>Ghana</td>
<td>Commercial charcoal</td>
<td>Locally made</td>
<td>2-8 t ha⁻¹</td>
<td>F</td>
<td>1 crop cycle</td>
<td>Oats</td>
<td>AC, sandy-loam</td>
<td>IN</td>
<td>+100% (fruit)</td>
</tr>
<tr>
<td>Kammann et al. (2015)</td>
<td>Germany</td>
<td>Wood chips, co-composted</td>
<td>PY, 700, co-composted</td>
<td>20 g kg⁻¹</td>
<td>GH</td>
<td>12 weeks</td>
<td>Chenopodium quinoa</td>
<td>Nutrient poor, sandy loam</td>
<td>IN, ORG</td>
<td>+200% (biomass) with co-composted BC, 40% with pure BC</td>
</tr>
<tr>
<td>Nielsen et al. (2014)</td>
<td>Australia</td>
<td>Jarrah wood</td>
<td>PY, 600 and activated</td>
<td>1.1 and 5.44 t ha⁻¹</td>
<td>F</td>
<td>1 crop cycle</td>
<td>Sweet maize</td>
<td>AC, Red Fernosol</td>
<td>0</td>
<td>Neutral</td>
</tr>
<tr>
<td>Liu et al. (2015)</td>
<td>China</td>
<td>Maize stalks</td>
<td>CA, 400</td>
<td>18 t ha⁻¹</td>
<td>P</td>
<td>1 year (2 crop cycles)</td>
<td>Soybean, wheat</td>
<td>Coastal saline (AK foamy sand)</td>
<td>IN</td>
<td>+11% (soybean), +28% (wheat)</td>
</tr>
<tr>
<td>Fox et al. (2014)</td>
<td>Ireland</td>
<td>Miscanthus</td>
<td>PY, 600</td>
<td>10 and 20 g kg⁻¹</td>
<td>P</td>
<td>18 weeks</td>
<td>Italian rye grass</td>
<td>Sub-AC, loam</td>
<td>0</td>
<td>+ (93-145)% (biomass)</td>
</tr>
<tr>
<td>Deenik et al. (2010)</td>
<td>USA</td>
<td>Macadamia nut shell</td>
<td>CA, 50-100</td>
<td>70, 100, and 200 g kg⁻¹</td>
<td>GH</td>
<td>5 weeks</td>
<td>Lettuce and cabbage</td>
<td>Andosol (pH 6.4)</td>
<td>0, IN</td>
<td>+ (18-23)% (lettuce), (25-50)% (maize)</td>
</tr>
<tr>
<td>Deenik et al. (2010)</td>
<td>USA</td>
<td>Macadamia nut shell</td>
<td>CA, 50-100</td>
<td>50 g kg⁻¹</td>
<td>GH</td>
<td>4 weeks</td>
<td>Maize</td>
<td>Ultisol (pH 4.7)</td>
<td>IN</td>
<td>+50% (biomass)</td>
</tr>
<tr>
<td>De La Rosa et al. (2014)</td>
<td>Spain</td>
<td>Wood, paper sludge and sewage sludge respectively</td>
<td>PY, 500-600</td>
<td>10, 20 and 40 g kg⁻¹</td>
<td>P</td>
<td>11 weeks</td>
<td>Italian rye grass</td>
<td>Calcium Cambisol (sandy loam, pH 6.4)</td>
<td>0</td>
<td>+ (0-33)% (biomass)</td>
</tr>
<tr>
<td>Baronti et al. (2014)</td>
<td>Italy</td>
<td>Orchard pruning</td>
<td>PY, 500</td>
<td>22 and 44 t ha⁻¹</td>
<td>F</td>
<td>3 years</td>
<td>Grape</td>
<td>AC, sandy-clay-loam</td>
<td>IN</td>
<td>+ (24-57)% (UAP)</td>
</tr>
<tr>
<td>Genesio et al. (2015)</td>
<td>Italy</td>
<td>Orchard pruning</td>
<td>PY, 500</td>
<td>22 and 44 t ha⁻¹</td>
<td>F</td>
<td>4 years</td>
<td>Grape</td>
<td>AC, sandy-clay-loam</td>
<td>IN</td>
<td>+ (16-60)% (grape)</td>
</tr>
</tbody>
</table>

Continued on next page.
distribution feasibility. Negative effects on crop growth are mostly reported with BCs obtained from municipal waste, food waste, and sewage sludge because their excessive Na contents increase soil salinity (Liu et al., 2013; Wisnubroto et al., 2011; Rajkovich et al., 2012). Other negative effects from plant- and wood-based BCs are due to one of the following causes: high application rates, high volatile matter contents detrimental to crop growth, reduced plant available N, or negative liming effect in alkaline and calcareous

**Table 2.** Continued from previous page.

<table>
<thead>
<tr>
<th>References</th>
<th>Location</th>
<th>Feedstock</th>
<th>Process condition*</th>
<th>Application rate†</th>
<th>Type of experiment</th>
<th>Duration</th>
<th>Crop grown</th>
<th>Soil type</th>
<th>Fertiliser</th>
<th>Effects on crop yield</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kammann et al. (2012)</td>
<td>Germany</td>
<td>Peanut hull</td>
<td>PY, 500</td>
<td>50 t ha−1</td>
<td>GH</td>
<td>20 weeks</td>
<td>Italian ryegrass</td>
<td>Sub-Ac, sandy-loam</td>
<td>IN, ORG</td>
<td>+20% (biomass)</td>
</tr>
<tr>
<td>Ukoma et al. (2011)</td>
<td>Japan</td>
<td>Cow manure</td>
<td>PY, 500</td>
<td>10, 15 and 20 t ha−1</td>
<td>GH</td>
<td>12 weeks</td>
<td>Maize</td>
<td>Sandy (pH 6.36)</td>
<td>IN</td>
<td>Neutral (10 t ha−1), + (30-50) (grain)</td>
</tr>
<tr>
<td>Tammeorg et al. (2014)</td>
<td>Finland</td>
<td>Spruce and pine chips</td>
<td>PY, 500</td>
<td>5 and 10 t ha−1</td>
<td>F</td>
<td>3 years</td>
<td>Faba bean, turnip rape and wheat</td>
<td>Sandy clay-loam (pH 6.6)</td>
<td>0, IN</td>
<td>Neutral</td>
</tr>
<tr>
<td>Herath et al. (2015)</td>
<td>Sri Lanka</td>
<td>Glicicida sepium</td>
<td>PY, 500</td>
<td>10, 25 and 50 kg ha−1</td>
<td>GH</td>
<td>9 weeks</td>
<td>Tomato</td>
<td>Serpentine soil (Inceptisol, pH 5.5)</td>
<td>0</td>
<td>+40% (biomass)</td>
</tr>
<tr>
<td>Bass et al. (2016)</td>
<td>Australia</td>
<td>Willow</td>
<td>PY, 600</td>
<td>10 t ha−1 (BC only)</td>
<td>F</td>
<td>1 crop cycle</td>
<td>Each crop</td>
<td>Banana, papaya</td>
<td>Red Ferralsol (pH 6.2) for banana and red chromosol (pH 6.8) for papaya</td>
<td>IN</td>
</tr>
<tr>
<td>Mandal et al. (2016)</td>
<td>Australia</td>
<td>Poultry litter</td>
<td>PY, 500</td>
<td>50 kg ha−1</td>
<td>P</td>
<td>5 weeks</td>
<td>Wheat</td>
<td>High OM soil (pH 8.3), Medium OM soil (pH 5.5)</td>
<td>IN, ORG</td>
<td>+20-57% (biomass)</td>
</tr>
<tr>
<td>Butta et al. (2015)</td>
<td>Thailand</td>
<td>Eucalyptus wood</td>
<td>Traditional Kin, 350</td>
<td>10, 20 and 40 kg ha−1</td>
<td>GH</td>
<td>13 weeks</td>
<td>Maize</td>
<td>Loamy-sand (pH 8.8) and silt clay-loam (pH 5.5)</td>
<td>IN</td>
<td>Neutral (first crop cycle), + (41-49%) (biomass) in second crop cycle</td>
</tr>
<tr>
<td>Butta et al. (2015)</td>
<td>Thailand</td>
<td>Eucalyptus wood</td>
<td>Flash CA, 800</td>
<td>10, 20 and 40 kg ha−1</td>
<td>GH</td>
<td>13 weeks</td>
<td>Maize</td>
<td>Loamy-sand (pH 5.5) and silt clay-loam (pH 6.9)</td>
<td>IN</td>
<td>-(7-24)% (first crop cycle), + (54-65%) (biomass) in second crop cycle</td>
</tr>
<tr>
<td>Houben et al. (2013)</td>
<td>Belgium</td>
<td>Miscanthus</td>
<td>PY, 600</td>
<td>10, 50 and 100 kg ha−1</td>
<td>GH</td>
<td>12 weeks</td>
<td>Rapsedeed</td>
<td>Sandy loam (pH 6.6)</td>
<td>IN</td>
<td>+(3-30)% fold (biomass)</td>
</tr>
<tr>
<td>Beesley et al. (2013)</td>
<td>Spain</td>
<td>Orchard pruning</td>
<td>PY, 500</td>
<td>30 cm³ 100 cm−3</td>
<td>P</td>
<td>4 weeks</td>
<td>Tomato</td>
<td>Mine soil (Arsenic contaminated, pH 5.0)</td>
<td>0, IN</td>
<td>Neutral</td>
</tr>
<tr>
<td>Schmidt et al. (2014)</td>
<td>Switzerland</td>
<td>Hardwood chips</td>
<td>PY, 500</td>
<td>8 t ha−1</td>
<td>F</td>
<td>3 years</td>
<td>Grape</td>
<td>Clay loam (pH 7.9)</td>
<td>ORG</td>
<td>Neutral</td>
</tr>
<tr>
<td>Gregory et al. (2014)</td>
<td>New Zealand</td>
<td>Willow wood</td>
<td>PY, 350 and 550</td>
<td>30 and 60 t ha−1</td>
<td>GH</td>
<td>6 months</td>
<td>Perennial ryegrass</td>
<td>Gravely soil (pH 5.8)</td>
<td>0</td>
<td>+(50-60)% (biomass)</td>
</tr>
<tr>
<td>Oram et al. (2014)</td>
<td>Netherlands</td>
<td>Natural grass cutting</td>
<td>PY, 400</td>
<td>10 t ha−1</td>
<td>GH</td>
<td>8 weeks</td>
<td>Red clover, red fescue and plantain</td>
<td>Podzol (pH 5.24)</td>
<td>0, IN</td>
<td>+(3-30)% in monocultures, + (28-50)% in mixed cropping</td>
</tr>
<tr>
<td>Jones et al. (2012)</td>
<td>UK</td>
<td>Mixed wood chips</td>
<td>PY, 450</td>
<td>25 and 50 t ha−1</td>
<td>F</td>
<td>3 years</td>
<td>Maize (year 1) and hay grass (years 2 and 3)</td>
<td>Sandy clay-loam (pH 6.7)</td>
<td>IN</td>
<td>Neutral (maize), +(13-32)% (hay)</td>
</tr>
<tr>
<td>Suddick and Sin (2013)</td>
<td>USA</td>
<td>Walnut shell</td>
<td>GA, 900</td>
<td>5 and 10 t ha−1</td>
<td>F</td>
<td>15 months</td>
<td>Lettuce, bell pepper and Swiss chard</td>
<td>Silty-loam (pH 7.6)</td>
<td>ORG, IN</td>
<td>Neutral</td>
</tr>
<tr>
<td>Gierefa et al. (2015)</td>
<td>Kenya</td>
<td>Sugarcane bagasse, Eucalyptus wood, Delonix regia, tea pruning, maine stover, maine cobs and rice hulls</td>
<td>PC, 150 or 550, pre-treated with acid, acetone and steam</td>
<td>15 t ha−1</td>
<td>GH</td>
<td>3 months</td>
<td>Common bean</td>
<td>Humic acrrol (pH 6.1)</td>
<td>IN</td>
<td>+16% (shoot, on average)</td>
</tr>
<tr>
<td>Rondon et al. (2007)</td>
<td>Colombia</td>
<td>Eucalyptus deglupta</td>
<td>CH, 350</td>
<td>40 and 90 g kg−1</td>
<td>P</td>
<td>11 weeks</td>
<td>Common bean</td>
<td>Clay-loam soil (pH 7.0)</td>
<td>IN</td>
<td>+39% (biomass), +46% (grain)</td>
</tr>
</tbody>
</table>

PP, pyrolysis; F, field; AC, acidic; IN, inorganic fertiliser; OA, pyrolysis; ORG, organic; GH, greenhouse; P, pot; K, nurses; UV, leaf water potential; CA, carbonisation; CH, charring; X, neutral pH. *Numbers refer to temperature in°C; †application rate expressed either in t ha−1 or mg kg−1 soil or cm³ cm⁻³ soil. +, increase; −, decrease (both compared to control).
soils (Deenik et al., 2010; Nelissen et al., 2014; Laghari et al., 2015; Bass et al., 2016; Marks et al., 2014).

Baronti et al. (2010), in a pot experiment with high BC rates, reported a perennial ryegrass yield increase by 29–120% after BC derived from orchard pruning was applied at between 30-60 t ha⁻¹ to sandy-loam soil with sub-acid pH. When the application rate was raised to between 100 and 120 t ha⁻¹, a 10-20% decrease in yield was observed. When Deenik et al. (2010) and Laghari et al. (2015) observed similar results, it suggested that higher dose application to soil might negatively affect crop yield ascribed to either immobilisation of N onto BC surfaces that makes them unavailable to plants, or release of a toxic or volatile compounds (e.g. PAHs, PCBs) detrimental to crop growth (Clough et al., 2013; Spokas et al., 2011).

A glance at other reviews

Several meta-analyses have shown that BC positively affects crop yields. One done by Jeffery et al. (2011), based on 26 case studies that included 177 treatments, indicated only a small (10%) overall productivity improvement on agricultural soils amended with BC. Another, compiled from 371 case studies published in 114 articles and performed by Biderman and Harpole (2013) showed that applying BC to agricultural soils, on average, increased aboveground biomass productivity, crop yield, nutrient availability, microbial biomass, and root nodulation in N-fixing crops; only a few limited studies showed negative crop effects (Jeffery et al., 2011). More recently, a weighted meta-analysis of 103 case studies on the effect of BC on crop productivity by Liu et al. (2013) demonstrated that BC application at rates below 30 t ha⁻¹ have significant and positive effects (productivity increased 11% versus control, on average). Liu et al. (2013) also noted that response varied as experimental conditions changed. For example, the greatest mean increase occurred with manure BCs. In addition, all three meta-analyses revealed the same phenomena about crop response to BC: greater in pot and greenhouse than in field experiments; greater in soil with acidic pH than in neutral and alkaline pH; greater in sandy textured soils than in silt and loam soils. The results also made clear two effects. First, there is a positive liming effect with increased nutrient availability that alkaline BCs bring to acidic soils (Ameeloot et al., 2013a, 2014; Fox et al., 2014; Srinivasan et al., 2015; Subedi et al., 2016a, 2016b), and second, BCs provide an improved soil structure with increased water holding capacity in sandy soils thereby reduced water stress and greater water use efficiency by the crops (Mukherjee and Lal, 2013; Baronti et al., 2014). Finally, each meta-analysis confirmed that BCs derived from manure, wood, and straw produced more consistent positive crop responses than did BCs derived from other feedstocks, although crop productivity effects might be overstated in pot or greenhouse experiments versus field experiments (Liu et al., 2013).

Potential risks associated with biochar application to agricultural soils

Risks associated with BC application to agricultural soils relate principally to its inherent properties that originate from feedstock type and pyrolysis conditions, and not to environmental conditions (Barrow, 2012; Kuppusamy et al., 2016). There are five main categories of risk: toxic compound and heavy metal release, heavy metal mobilisation, pesticide/other compound retention, and dust production during application.

Biochar as a source of toxic compounds and heavy metals

Biochar prepared from different feedstocks can contain sizeable quantities of volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), and polychlorinated benzenes (PCBs) in various concentrations. Whether or not VOCs stimulate or suppress plant and microbial growth depends on their type and composition (Deenik et al., 2010; Spokas et al., 2011). PAHs and PCBs are phytotoxic to seed germination and plant growth, and may also be ecotoxic (Quilliam et al., 2013b; Butnan et al., 2015; Domene et al., 2015; Genesio et al., 2015; Rombolà et al., 2015). However, concentrations of such compounds are generally either negligible or below thresholds set by BC certification standards (Appendix Table 2; EBC, 2012; IBI, 2014). No clear evidence links the presence of these compounds to feedstock type. Instead, their presence is strongly linked to process conditions (pyrolysis, gasification, hydrothermal carbonisation) (Spokas et al., 2011; Freddo et al., 2012; Hale et al., 2012; Subedi et al., 2015; Camps-Arbestain et al., 2014). In general, fast pyrolysis increases the risk of generating such toxic compounds. Therefore, general pyrotoxicity and ecotoxicity tests are recommended before BC agricultural use (Domene et al., 2015; EBC, 2012; Montanarella and Lugato, 2013).

Furthermore, heavy metals can accumulate in BCs derived from manure, sewage sludge, municipal waste, and biosolids, therefore, represent another concern that engineers and agronomists need to address (Kloss et al., 2014; Fellet et al., 2011, 2014; Camps-Arbestain et al., 2014). In fact, the process of BC production can actually concentrate heavy metals naturally present in the feedstock (Table 1); the need to distribute a large quantity of stable C raises the risk of distributing high amounts of heavy metals. Subedi et al. (2016a), Singh et al. (2010), Cantrell et al. (2012), Hossain et al. (2015), and Bachmann et al. (2016) have all reported high Zn (>1200 mg kg⁻¹), Mn (>1000 mg kg⁻¹), and Cu (>360 mg kg⁻¹) concentrations in poultry litter, swine manure, sewage sludge, and Eucalyptus-derived BCs (Appendix Table 2). These elemental concentrations generally increase as PTs rise and are concentrated mainly in the ash fraction (Domene et al., 2015). However, in general, most authors have reported BC heavy metal concentrations at levels below the limit set for quality standards (EBC, 2012; IBI, 2014), which deems them suitable for land application (Srinivasan et al., 2015). Such BCs, irrespective of their positive crop effects, demand further investigation, as they may represent a risk to soil microbe, human, and livestock health (Kuppusamy et al., 2016).

Retention of organic and inorganic compounds

Some substances can be sorbed onto BC surfaces and consequently their bio-availability can be reduced. Soil-applied pesticides can be sorbed onto the surface of BC. This reduces leaching, and at the same time efficacy, as it limits both biodegradation and plant uptake (Jones et al., 2011; Camps-Arbestain et al., 2014; Kuppusamy et al., 2016). This eventually increases pesticide residual life in soil and negatively affects soil micro-biota. Similarly, absorption capacity of BC has a potential to mitigate the bioavailability of heavy metals in contaminated soils (Beesley et al., 2010; Domene et al., 2015; Fellet et al., 2011, 2014; Herath et al., 2015; Hossain et al., 2015).

Clearly, the capacity of BC to adsorb a range of contaminants (both organic and inorganic) may lead to an imbalance in the uptake of plant nutrients and may affect product quality (Camps-Arbestain et al., 2014; Kuppusamy et al., 2016). These less-explored aspects of BC soil application need further attention.
Mobilisation of heavy metals in soils

Contrary to immobilisation of heavy metals, some authors have reported increased mobilisation in BC-amended soils posing further risks. Beesley et al. (2010, 2013) have reported increased Cu and As availability in soil amended with BCs derived from hardwoods and orchard prunings; Uchimiya et al. (2010) suggested that this could be due to a mobilisation of soil-retained metals by dissolved organic C (DOC). Mobilisation of heavy metals is therefore another potential threat to plant and animal health.

Production of dust

The dusty nature of BC may pose yet another risk to human health if inhaled during application. The problem is particularly relevant for BCs derived from rice and wheat husks, cow manure, and so on (Camps-Arbestain et al., 2014). Health risks from BC contaminants (PAHs, PCBs and heavy metals) have been previously described.

Moreover, BC application to soil is thought to alter Earth’s surface albedo, which may negatively interfere with its climate change mitigation potential (Meyer et al., 2012; Montanarella and Lugato, 2013). Meyer et al. (2012) reported that wood BC applied in a test field at 30–32 t ha⁻¹ reduced its global warming mitigation benefit by 13–22% due to a change in albedo (relative to an analysis that disregarded the albedo effect). A very recent study on the atmospheric effects of BC warned that black-C aerosol from BCs might threaten its negative emission potential due to its unavoidable fragmentation into tiny particles (<2.5 μm), possibly during its production, storage, transportation, and application (Genesio et al., 2016). These findings draw attention to the use of BC as a tool for climate change mitigation, and certainly require further investigation.

Conclusions

Applying BC to soil achieves multiple benefits (C sequestration, GHGs mitigation, soil fertility improvement, and increasing crop productivity), but the benefits vary considerably with BC type and application rate, elapsed time from application, crop type, and climatic conditions. Its production from locally available waste biomasses (crop/wood residues, manure, sludge, and so forth) is determined by feedstock type and climatic conditions. Its production from locally available waste biomasses is richer in plant-available nutrients than wood-based BCs, despite the unavoidable losses of C, N, and S at high PTs. The literature suggests that BC has the potential to increase crop productivity, with more positive effects in acid soils than in alkaline soils, and are subject to further risks. Beesley et al. (2010) suggest that disregarded the albedo effect). A very recent study on the atmospheric effects of BC warned that black-C aerosol from BCs might threaten its negative emission potential due to its unavoidable fragmentation into tiny particles (<2.5 μm), possibly during its production, storage, transportation, and application (Genesio et al., 2016). These findings draw attention to the use of BC as a tool for climate change mitigation, and certainly require further investigation.

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