Biochar composition-dependent impacts on soil nutrient release, carbon mineralization, and potential environmental risk: A review

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HIGHLIGHTS

- Nutrient contents in biochar highly dependent on the feedstock type.
- Pyrolysis temperature alters the proportion of aromatic and aliphatic C fractions.
- Chemical and physical properties of biochar affect the nutrient release from biochar.
- Application of unsuitable biochar can negatively affect environmental quality and human health.
- Biochar can be a potential source of polycyclic aromatic hydrocarbons and polychlorinated dibenzodioxins.

ABSTRACT

Biochar application has multiple benefits for soil fertility improvement and climate change mitigation. Biochar can act as a source of nutrients and sequester carbon (C) in the soil. The nutrient release capacity of biochar once applied to the soil varies with the composition of the biochar, which is a function of the feedstock type and pyrolysis condition used for biochar production. Biochar has a crucial influence on soil C mineralization, including its positive or negative priming of microorganisms involved in soil C cycling. However, in various cases, biochar application to the soil may cause negative effects in the soil and the wider environment. For instance, biochar may suppress soil nutrient availability and crop productivity due to the reduction in plant nutrient uptake or reduction in soil C mineralization. Biochar application may also negatively affect environmental quality and human health because of harmful compounds such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated dibenzodioxins, and dibenzofurans (PCDD/DF). In this review, we discuss the linkage between biochar composition and function, evaluate the role biochar plays in soil fertility improvement and C sequestration, and discuss regulations and concerns regarding biochar's negative environmental impact. We also summarize advancements in biochar production technologies and discuss future challenges and priorities in biochar research.

Keywords: Black carbon; Carbon storage; Policy development; Priming effect; Soil nutrients availability

1. Background

The United Nations Sustainable Development Goals (SDGs) emphasize soil fertility improvement and C sequestration as one of the SDGs, and propose reasonable targets for nations to achieve by 2030. The SDGs highlight the necessity of soil security by improving its fertility to supply plants with sufficient and balanced nutrients. Maintaining good soil physical, chemical, and biological properties is essential to ensuring soil security, sustaining high crop yield, and improving rural economy (Adhikari and Hartemink, 2016). Recently, an increasing emphasis has been given to the restoration and rehabilitation of low-fertility and degraded soils to achieve the potential maximum production rate to meet the growing demand for food by the burgeoning world population (Lal, 2015; Beiyuan et al., 2016; León et al., 2017).

Soil C storage is an important indicator of soil fertility and health, as it plays a vital role in different biogeochemical processes in the soil (Doetterl et al., 2016). Considerable attention has been given to tackle soil C loss in the form of CO_2. In the last two decades, anthropogenic CO_2 emissions have increased by more than 3% annually, thereby threatening various ecosystems on the earth (Woolf et al., 2010). The rising atmospheric CO_2 concentration is triggering an alarming increase in global temperature and causing extreme weather events, such as droughts and floods, leading to desertification, declining glacial area, and unprecedented sea-level rise (Hansen et al., 2017). Applicable strategies of climate change mitigation, including the rapid phasing out of fossil fuel use, enhancement in soil C sinks and deployment of feasible CO_2 removal approaches, are urgently needed to overcome this threat to mankind (Von Stechow et al., 2015; Fellmann et al., 2018). Carbon sequestration in soils is a viable approach to compensate for the increased CO_2 efflux from soils (Lal et al., 2015; Awad et al., 2017; Minasny et al., 2017).

Different management strategies have been applied to improve soil fertility and mitigate climate change. Conventional organic soil amendments, including animal manure, sewage sludge, mulches and composts, have been used for such purposes (Lal, 2004; Stefaniuk et al., 2018). However, most of these management approaches make limited or no contribution to C storage in soils due to the fast decomposition of organic carbon (OC), thereby resulting in CO_2 emissions and loss of their efficacy in maintaining the C balance in the soil (Lehman, 2007; Schmidt et al., 2011; Paustian et al., 2016; Agegnehu et al., 2017). Moreover, manure, sewage sludge, and composts may contain pathogens, potentially toxic metals, and harmful pharmaceutical compounds (Verlicchi and Zambello, 2015). These components may cause soil contamination in the long-term. Soil application of composts and manures may also contribute to excessive nitrate concentration in soils and increased emissions of nitrous oxide, ammonia, and methane, which could pollute the groundwater and surface water and contribute to global warming (Ding et al., 2016; Van Groenigen et al., 2017).

Since the green revolution, inorganic fertilizers have been widely applied to soils to increase soil productivity (Vanlauwe et al., 2010). However, intensive agricultural practices with sole reliance on inorganic fertilizers are usually costly and detrimental to soil quality and ecosystem health (Karer et al., 2015; Ding et al., 2016; Srinivasarao et al., 2014; Carlson et al., 2015). Consequently, it is imperative to employ eco-friendly and pragmatic alternate approaches to improve soil fertility (Inyang et al., 2015; Ok et al., 2015). In the last two decades, biochar has received growing interests for its application to soil due to its multiple benefits for soil quality improvement, waste management, energy production, and climate change mitigation (Usman et al., 2016; Awad et al., 2018; El-Naggar et al., 2018a,b). Biochar is a carbonaceous material produced by pyrolysis of biomass waste (Lehmann and Joseph, 2009). It is a promising and cost-effective strategy to improve soil fertility and simultaneously sequester C in soils (Ahmad et al., 2016; Igalavithana et al., 2016; Smith et al., 2016; Hussain et al., 2017).

Recent studies on the impact of biochar on soil quality, however, have reported contrasting results showing positive, negative, or neutral effects (Beiyuan et al., 2017; Igalavithana et al., 2018; Yang et al., 2019). For instance, biochars derived from different feedstocks (wood, rice straw, and grass residues) display different potentials to improve the fertility of two soils (sandy and sandy loam) in an incubation experiment (El-Naggar et al., 2018c), where the application of rice straw biochar significantly increased the contents of N, available P, and exchangeable cations, and enhanced the CO_2 efflux as compared to wood and grass biochars in the sandy soil. In a greenhouse experiment with biochars produced from five different feedstocks, the results were strongly dependent on the biochar type (Alburquerque et al., 2014). For example, wheat straw and olive tree pruning-derived biochars increased the soil dissolved OC, while olive stone, almond shell, and pine wood chip-derived biochars had minimal effect on soil dissolved OC. The authors also reported that soils treated with wheat straw and pine wood chip biochars exhibited greater field capacity than soils treated with other types of biochars. The contradictory results of these studies can be partly attributed to factors such as the soil type and experimental setup. However, one of the most important reasons for the contrasting performance of the biochars is the different composition of each biochar type. Each biochar produced from a specific feedstock using a specific production method (e.g., pyrolysis, gasification, and hydrothermal carbonization) using a specific temperature and with/without an activation or modification process will yield a unique biochar material (Igalavithana et al., 2017a; Yoo et al., 2018; You et al., 2017, 2018; El-Naggar et al., 2019). Taking this fact into account, it would be problematic to generalize the role of biochar in different applications without defining the production conditions and biochar composition.

Some review papers have documented variations in biochar properties and functions in soil based on feedstock type and production condition (e.g., Khura et al., 2015; Xie et al., 2015; Ding et al., 2016; Agegnehu et al., 2017; Igalavithana et al., 2017a). However, to our knowledge, none of the current literature has highlighted their important effects on soil quality as the main focus. Therefore, in the current review, we aim to elucidate the biochar composition-dependent impact in three main areas: nutrient content and release, C sequestration and dynamics, and the potential negative impact on the environment.

2. Biochar application to improve soil fertility

The application of biochar can enhance soil water availability (Ma et al., 2016), water holding capacity (Mohamed et al., 2016), soil aeration (Cayuela et al., 2013), soil organic carbon (SOC) content (El-Naggar et al., 2018b), soil microbial biomass and activity (Igalavithana et al., 2017b), enzymatic activity (Awad et al., 2018), and nutrient retention and availability (El-Naggar et al., 2015, 2018a,b), which result in less fertilizer needs and reduce nutrient leaching (Lehmann et al., 2003). A summary of the impact of biochar application on soil properties is presented in Table 1. Although many studies showed the efficacy of biochar as a soil amendment (Table 1), some studies reported decreasing crop productivity after biochar application (Schmidt et al., 2015), which could be related to reduction in plant nutrient uptake or reduction in soil C mineralization (Ippolito et al., 2012). These contradictory results on crop yield in biochar-amended soils were likely due to the variability in biochar and soil properties. For example, biochar produced at high pyrolytic temperatures ($\geq$ 600 °C) may adsorb plant nutrients, thereby restricting plant uptake. In addition, the negative priming effect (PE) induced by nutrient adsorption by biochar may also cause a reduction in nutrient availability for plant uptake in soils containing low OC (Kuppusamy et al., 2016). Therefore, these two key factors (nutrient content of biochar and induced PE) need to be further studied when investigating the impact of biochar on soil fertility.

3. Biochar as a source of available nutrients

3.1. Effects of feedstock type and pyrolysis methodology on nutrient content in biochar

Biochar could be a valuable source of nutrients for plants if the pyrolysis process is managed to preserve the nutrients. The total nutrient content of biochar is not only a function of feedstock composition, but also a function of many different factors, including pyrolysis temperature, duration, and gaseous environment (e.g., CO_2, N_2). The influence of feedstock type and pyrolysis temperature on biochar properties has been documented from a large number of biochar studies (Figure 1). The nutrient contents in biochar are highly dependent on the feedstock type. For instance, the N and P contents are usually higher in biochars produced from manure, followed by those produced from grass and wood, while C content is usually higher in biochars produced from wood than those produced from grasses, followed by manure (Figure 1). Several types of feedstock have been used for biochar production. In general, organic wastes with rich nutrient contents produce biochars with a higher nutrient content (Table 1). Figueredo et al. (2017) found that biochar produced from sewage sludge at 350 °C had a higher N content (3.17%) compared to that produced from sugarcane and eucalyptus wastes (1.4 and 0.4%, respectively). In another study, pyrolysis of swine wastes increased N and P concentrations from 1.8 and 1.6% in the raw swine solids to 2.1 and 3.8% in the biochar produced at 420 °C, respectively, while the biochar produced from wood chips under the same conditions contained less N and P (1 and 1.3%, respectively) (Marchetti and Castelli, 2013).

The increase in nutrient concentrations in the biochar as compared to that in the raw feedstock is mainly due to the weight loss during pyrolysis. Thus, nutrients become enriched in the biochar as compared with the feedstock, even though a significant portion of the biomass is lost during biochar production. For instance, in the previous study (Marchetti and Castelli, 2013), the total N content decreased by 58% in the swine waste biochar and by 53% in the wood chip biochar, while the total P content decreased by 17% and 27% in the swine waste and wood chip biochars, respectively. Nitrogen loss during pyrolysis was attributed to the volatilization of NH_4^+. Similarly, Hass et al. (2012) observed that chicken manure-derived biochar at 350 °C recovered 57% of the original dry mass as compared to 38% at 700 °C. In the same study, a large portion of the C and N was lost during pyrolysis. The preferential volatilization of N over C resulted in an increase in the C/N ratio of the biochar with increasing temperature. The total N, P, and K contents of biochar produced from chicken manure at 350 °C was 38, 27, and 56 g kg^{-1}, respectively (Hass et al., 2012). Increased pyrolysis temperature and activation could decrease the macro- and micronutrient contents and their availability to plants following soil application of biochar. Sahin et al. (2017) indicated that acid activation of biochar reduced its N and micronutrient contents. Borchard et al. (2012) found that the physical activation of biochar decreased the contents of available NO_3^--N and P by about 55 and 90% (w/w), respectively. The loss of available N was attributed to the release of volatile N-containing compounds during the activation process and to the net transfer of labile N into heterocyclic N forms (Borchard et al., 2012).

3.2. Relationship between biochar chemical composition and nutrient release

The total nutrient content in biochar does not necessarily reflect the release of all nutrients from biochar when it is applied to the soil. Nutrients, especially N, in biochar tend to be less available compared to those in the original feedstock. For instance, El-Naggar et al. (2015) found that only 4.5% of the N content of the added wood biochar was turned into soil-available N compared to 15.6% for the N in the original feedstock. The high C/N ratio of biochar, and N enmeshment in the stable biochar material would result in N immobilization. This might be the reason for the insignificant contribution of biochar to the N budget of crops (Asai et al., 2009; Hangs et al., 2016; Nguyen et al., 2017). In a short-term experiment, Nelson et al. (2011) suggested the need for N fertilization in addition to biochar application in order to improve the N status in biochar-amended soils.

In a batch extraction and column leaching experiment, Mukherjee and Zimmerman (2013) determined nutrient release from a variety of new and aged biochars to solution (Figure 2). Different biochar samples, except for N-rich biochars, exhibited minor N release after successive batch extractions. The nutrient release from biochar to solution varied with feedstock type. Ammonium is the major form of N released from biochar, followed by organic N, while nitrate ranged between 2% and 30% in the leachates, while organic N was up to 59%. The release of dissolved OC, N, and P into the soil solution was significantly correlated with biochar volatile matter contents and acid functional group density (Mukherjee and Zimmerman, 2013).

The release of nutrients from biochar to soil solution differs from one element to another depending on the sorption affinity of the individual element with the biochar and/or the soil. Angst and Sohi (2013) conducted a sequential leaching experiment with deionized water to study nutrient release from hardwood biochars. They found that P release decreased gradually, where the sixth extraction yielded 44–73% P in comparison with the first extraction. Similarly, K release was higher at the beginning and declined rapidly, where the sixth extraction yielded only 6 to 18% K as compared with the first extraction. In comparison to rapid K release, the gradual release of P from biochar suggested a sustainable gradual supply throughout the crop-growing season. Therefore, the differences in the release patterns of individual nutrient elements and the type of crops concerned should be considered when managing crop nutrient supply with the application of biochar.

3.3. Relationship between physical properties of biochar and nutrient release

The physical properties of biochar are a function of production conditions (Kim et al., 2012). For instance, the surface area of mulberry wood biochar increased from 16.5 to 58.0 m^2 g^{-1} when the pyrolysis temperature increased from 350 to 550 °C, respectively (Zama et al., 2017). The feedstock type also plays an important role in determining the physical properties of biochar. For instance, the surface area of oak bark-derived biochar was greater than that of oak wood-derived biochar (8.8 m^2 g^{-1} and 6.1 m^2 g^{-1}, respectively) (Mohan et al., 2014). The biochar produced from hardwood jarrah had greater microporosity than the softwood pine biochar (Shaheen et al., 2018). The disparities in the biochar physical properties from different feedstocks might be due to the varied contents of lignin, hemicellulose, and cellulose. This variation in biochar physical properties affects the functions of biochar in soils, including the retention/release of soil nutrients.

In an incubation experiment, biochars produced from vegetable waste and pinecone residues at different pyrolysis temperatures (i.e., 200 and 500 °C) were applied to contaminated soils at 5% (w/w) rate (Igalavithana et al., 2017b). The two biochars produced at 200 °C increased the size of the microbial communities, while the biochars produced at 500 °C suppressed the microbial communities in the soils. This was mainly attributed to the fact that the biochars produced with a lower pyrolysis temperature (200 °C) had higher volatile matter contents and lower resident material (lower structural stable C) than those produced with a higher pyrolysis temperature (500 °C); thus, the biochars pyrolyzed at 200 °C supplied the microbes with labile components through the readily released nutrients.

Weathering of biochar surfaces and pore edges in soil might also enrich the biochar surfaces with more oxidized functional groups and facilitate biochar-soil mineral interactions (El-Naggar et al., 2018b). In a field experiment, the particulate organic matter fraction of biochar had physical interactions with soil minerals in the coarse sand fraction, while the biochar formed organo-mineral complexes with soil minerals in the clay/silt fraction, because the clay/silt fraction of soil had higher exchangeable cations (e.g., Ca, Mg, Na and K) than the coarse sand fraction (El-Naggar et al., 2018b). Taherymoosavi et al. (2018) observed physical interactions on the surfaces of biochar produced at 450 °C between C and elements (Na, Ca, Mg, K, and Al) originated from mineral phyllosilicates. They also reported that the addition of basalt with wheat straw biochar produced at 550 °C led to the formation of organo-mineral complexes with the basalt minerals (e.g., Si, Al, K, and O) on the biochar surfaces (Figure 3), which protected the biochar surface from oxidation (as revealed by X-ray photoelectron spectroscopy results) more than that of wheat straw biochar having no such complexes on its surface. In the same study, wheat straw biochar with basalt produced at 650 °C was also examined. The scanning electron micrograph images and EDS mapping revealed that the biochar macropores were filled with minerals of basalt (e.g., Si, Al, K, and O) (Figures 4 and 5), thereby confirming the existence of physicochemical interactions within the porous structure of biochar. The organo-mineral complexes, coating, and pore interactions of biochar with minerals of soil or other amendments strongly affect the dynamics of releasing/retaining nutrients in soils. However, this area needs more investigation using integrated spectroscopic techniques to elucidate all related mechanisms and effects on soil nutrients.

4. Biochar application and soil carbon

4.1. Biochar as a source and sink of carbon

Carbon sequestration in soil is one of the principal strategies to combat climate change that is caused by anthropogenic CO_2 emissions (Paustian et al., 2016). Cultivation of cover crops is one of the conventional approaches to sequester C from the atmosphere, as plants sequester CO_2 in their biomass, which is then transferred to the soil in the form of organic matter (Lackner et al., 2003). The addition of plant residues to soil also plays a vital role as a source of C in the soil. However, the turnover of these organic materials is usually fast due to their fast decomposition rate; thus, the C added to the soil is quickly released back to the atmosphere. Converting plant residues into biochars through pyrolysis transforms the C into a more stable and recalcitrant form that could remain in the soil for thousands of years (Lehmann et al., 2007). Thus, biochar is considered not only a C source, but also a C sink in the soil (El-Naggar et al., 2018b). With biochar, annual net emissions of CO_2 could be offset by a maximum of 0.21 Pg CO_2-C equivalent, which is equal to about 12% of current anthropogenic CO_2-C emissions (Woolf et al., 2010).

Biochar is a C-rich material; however, the C contents in biochar vary mainly with feedstock type and pyrolysis temperature (Usman et al., 2015; El-Naggar et al., 2018c). For instance, biochar produced from wood biomass usually shows higher C contents than that produced from rice straws and crop residues (El-Naggar et al., 2018c). The C stability in biochar varies with feedstock type; for instance, wood biochar usually shows higher stability in soil than rice residue-derived biochar (El-Naggar et al., 2018c). The higher lignin content in wood biomass compared with that in crop residues contributes to the greater C stability in wood-derived biochar (Bird et al., 1999). Pyrolysis temperature is another critical factor that affects the C stability in biochar because it alters the proportion of aromatic and aliphatic C fractions, as well as the condensation of aromatic C in biochar (Kloss et al., 2012; Usman et al., 2015). Biochar produced under high pyrolysis temperatures usually contains more aromatic C than that produced under low pyrolysis temperatures. Thus, biochar produced under high pyrolysis temperatures is less degradable in soil than a low pyrolysis temperature product. Biochar stability in the soil is of paramount importance for its role in improving and maintaining soil properties relevant to crop production. Once applied to the soil, biochar stability determines the period over which the biochar product impacts C sequestration and climate change mitigation, as well as soil fertility improvement.

4.2. Biochar and soil carbon mineralization: positive or negative priming effect

Soil priming is known as the change in the decomposition rate of SOC following the addition of fresh organic amendment into the soil as compared with soil without amendment addition (Kuzyakov et al., 2000). The PE is a term that refers to the acceleration or inhibition of the rate of organic matter mineralization as a result of applying amendments (Gontikaki et al., 2013; Xu et al., 2018a). The prediction of PE following the addition of soil amendments is of great importance to understand the dynamics of SOC and the influence of different amendments on soil C stock and mineralization.

The application of biochar to soil was found to affect the mineralization of SOC in the long-term, thereby leading to a positive or negative PE in the soil (Figure 6) (Zimmerman et al., 2011; El-Naggar et al., 2018c). Whether biochar causes a positive or negative PE is still under debate (El-Naggar et al., 2015, 2018c; Xu et al., 2018a). One could hypothesize that biochar induces a negative PE when it is applied to the soil because biochar is highly porous in nature, which imparts its strong affinity for organic matter (Zimmerman et al., 2011). Biochar may sequester native soil organic matter within its pore network, thereby reducing the degradability of the organic matter in soil via microbial decomposition (Zimmerman et al., 2011). In contrast, biochar may also stimulate soil C mineralization, which is known as a positive PE (Luo et al., 2017). Biochar might provide a suitable habitat for microorganisms by supplying them with labile C, N, P and micronutrients, thereby improving the microbial growth and proliferation (Chan and Xu, 2009). This act might enhance the microbial activity and induce a positive PE in the soil (Figure 7).

The governing factors of biochar-induced PEs in soil include abiotic factors, such as soil moisture content, texture, clay content and SOC content, and biotic factors, such as fungi/bacteria composition and the abundance of saprophytic fungi and soil animals (Wang et al., 2016). The influence of these factors on inducing PE in soil depends on the initial soil properties and biochar feedstock type (El-Mahrouky et al., 2015). In a long-term incubation experiment, three types of biochars (rice straw, umbrella tree wood, and grass) were applied at 30 t h^{-1} to two types of soils (a sandy and a sandy loam soil). The results showed that the sandy loam soil had 2–3 times higher CO_2 emissions than those of the sandy soil due to the higher microbial community abundance in the sandy loam soil (Figure 8; El-Naggar et al., 2018d). In the study, different types of biochar did not significantly influence the soil PE in the sandy loam soil, but induced a positive PE in the sandy soil. The rice hull biochar treatment induced the highest rate of CO_2 emission, which was attributed to its high aliphatic dissolved OC content as compared to that of biochars produced from wood and grasses. Wang et al. (2016) conducted a meta-analysis based on 116 observations to estimate the PEs following biochar addition to soil. They reported that biochar commonly showed a negative PE in the soil (-3.8%) as compared to soils without biochar addition. In this meta-analysis study, sandy soils usually showed a positive PE following biochar addition (20.8%) due to the stimulation of microbial activities in soils with a poor soil fertility.

The above discussion indicates that there is still a lack of understanding in terms of the plausible impact of biochar on the PE of soil C, which warrants further studies involving biochar produced from various feedstock types and under different soil and crop types. Previous reports have suggested that biochar could remain in the soil on a centennial scale, and that it has many direct and indirect impacts on soil organic matter dynamics and C sequestration.

5. Limitations and concerns of using biochar as a soil amendment

Since the potential use of biochar for environmental protection and agricultural production has been realized (Lehmann, 2007), biochar has been produced from a wide range of biomass feedstock types using different pyrolysis procedures (Zhao et al., 2013; Ahmad et al., 2014; Mohan et al., 2014). The biochar industry and market are growing worldwide (Jirka and Tomlinson, 2013), therefore, some key issues need to be considered when biochar is applied to agricultural systems. These concerns are mainly related to the negative impact that biochar might impart on soil fertility and plant nutrition, or the occurrence of accompanying compounds that are potentially harmful to human health and the environment.

5.1. Potential negative impacts of biochar on nutrient availability and crop yield

Although most literature reported direct or indirect positive effects of biochar on soil nutrient availability, several reports showed that biochar applications could reduce the availability of some nutrients, thereby resulting in a yield reduction (Hussain et al., 2017). In a laboratory experiment, high rates of biochar application of over 1.7% (over 60 t ha^{-1}) caused a decline in perennial ryegrass dry matter production (Baronti et al., 2010). The decline was attributed to the modification of soil chemical and physical properties under high rates of biochar application. Mikan and Abrams (1995) reported the failure of woody plants to establish and survive due to the large accumulation of charcoal and deficiency of micronutrients caused by increased soil pH from soil biochar application. Similarly, Karer et al. (2013) indicated that although wood-based biochar improved the water holding capacity in a Cambisol, its contribution to the macro- and micronutrients supply to crops was inhibited. A negative impact of biochar on yield and nutrient uptake was observed when biochar was applied at a rate of 72 t ha^{-1}, where maize and wheat grain yields decreased by 46 and 70%, respectively. The decrease in yield was attributed to the immobilization of N and micronutrients, which reduced their availability to plants under increased pH conditions. Bruun et al. (2012) compared different biochars produced at different fast and slow pyrolysis conditions and studied their effects on soil C and N dynamics. They found that the application of biochars produced with fast pyrolysis from wheat straw immobilized 43% of the inorganic N during 65 days of incubation, while biochars produced through slow pyrolysis increased the N mineralization rate by 7%.

In general, these results suggest that biochar could be a useful material for environmental management and agricultural production if an accurate application rate of biochar produced from appropriate feedstock using suitable pyrolysis technology is applied to the soil. As biochar application is a relatively new agricultural practice, there is a scarcity of field data about the long-term effect of biochar on the soil chemical, physical, and biological properties. There is also limited knowledge about the sustainability of biochar use for agricultural production, especially for the recommended annual biochar application rates in long-term and different cropping systems and its subsequent impact on nutrient availability and inherent soil fertility. We need to study and determine the maximum amount of biochar that can be applied to the soil (e.g., over several applications over several years) before the applied biochar begins to cause negative effects on nutrient availability and plant productivity.

5.2. Biochar regulations and concerns regarding potential environmental risks

Biochar can potentially be used for the treatment and restoration of infertile soils that are contaminated with various pollutants, such as potentially toxic metals (Beesley et al., 2011; Mandal et al., 2017a; Xu et al., 2018b), polychlorinated biphenyls (PCBs) (Denyes et al., 2012), pesticide residues (Zheng et al., 2010; Mandal et al., 2017b), and polycyclic aromatic hydrocarbons (PAHs) (Stefaniuk and Oleszczuk, 2016). Although biochar was found to be useful for immobilizing soil pollutants (Stefaniuk et al., 2017), several studies reported that some biochar products and production methods increased the availability of harmful organic compounds, which might represent a potential source of hazards to human health. For instance, Lyu et al. (2016) found that biochar could be a potential source of contaminants, particularly for PAHs and PCDD/DF, which could be generated during the pyrolysis or gasification process. Kookana et al. (2011) reviewed the potential unintended consequences of biochar, and reported that residues of some pollutants (e.g., PAHs, cresols, xylenols, formaldehyde, acrolein, etc.) could accumulate in biochar and pose a risk to microorganisms, plants and soil health. However, the content of those organic toxicants in the biochar and their ecotoxicological impacts on soil flora and fauna are not well documented (Kookana et al., 2011).

The production condition of biochar including the residence time during the pyrolysis specifically appears to be responsible for influencing the PAH concentrations in biochar. Brown et al. (2006) analyzed the concentrations of PAHs in biochars produced in a range of pyrolysis temperatures (450-1000 °C). They reported that PAH concentrations in biochar strongly depend on the production temperature of the material. Higher concentrations of low molecular weight PAHs were found in the biochars produced at low temperatures, while higher concentrations of high molecular weight PAHs were found in the biochars produced at high temperatures (Brown et al., 2006). Moreover, the pyrolysis process (slow or fast) plays a major role in determining the content and type of PAHs in biochar (Wang et al., 2017). Slow pyrolysis and long residence time was found to result in lower PAH yields than fast pyrolysis and short residence time (Wang et al., 2017).

In a greenhouse experiment, kiln wood biochar application increased the content of PAHs by 10 times in soils (José et al., 2016). This increase in the PAH content was attributed to the usage of traditional kilns in which syngas and tar oils are not removed. The use of modern gasification reactors to remove or capture syngas and tar oils could potentially address this issue of PAHs in biochar produced in kilns (José et al., 2016). This is in agreement with Garcia-Perez et al. (2008), who reported that PAHs escape with the gas during slow pyrolysis. Therefore, different organizations set threshold values for PAHs in biochar. The International Biochar Initiative set 6–20 mg kg^{-1} as the threshold value for the total concentration of 16 PAHs that were reported as toxic by the EPA (IBI, 2012). The European Biochar Foundation similarly set values of 12 mg kg^{-1} dry matter (DM) for basic grade biochar and under 4 mg kg^{-1} DM for premium grade biochar (EBC, 2013). Wang et al. (2017) reported that PAH concentrations showed a wide variation from less than 0.1 mg kg^{-1} to more than 10,000 mg kg^{-1} in various biochar products. This is why special care should be taken to decide the pyrolysis process and intended characteristics of the produced biochar before its application to agricultural soils.

6. Advancements in biochar production for soil fertility improvement and soil carbon sequestration

The chemical and physical properties of biochars depend on the production condition and feedstock type (Novak et al., 2009; Al-Wabel et al., 2013). The potential of biochar to improve the fertility of soils differs accordingly. There is a growing interest in improving biochar efficacy to promote soil fertility and soil C storage by applying advanced technology in the biochar production process. Products of these types of modification processes are known as designer/engineered biochar (Mandal et al., 2016; Rajapaksha et al., 2016). Designing the appropriate biochar (with desired properties) for the appropriate soil (with specific soil quality issues) is a promising strategy in the field of biochar application to soil (Novak et al., 2009; Atkinson et al., 2010; Singh et al., 2010; Abiven et al., 2014). This strategy can be developed by designing or modifying biochar through physicochemical alterations or controlling the pyrolytic process. These modification methods include co-composting biochar with organic or composted materials.

Adding biochar to the composting process can stimulate the process and enhance the quality of the end product (co-composted biochar). The benefits of biochar addition to the compositing process include stimulating microbial activity, improving the C/N ratio, maintaining the temperature and homogeneity of the mixture, and enhancing the product's organic matter content (Prost et al., 2013; Zhang and Sun, 2014). It could also enhance the structure of the compost and reduce nutrient loss. At the same time, the composting process will also enhance the biochar properties, such as charging its surface with nutrients. The potential of co-composted biochar to improve soil fertility and soil C sequestration has been reported (Khan et al., 2014). For instance, the application of co-composted biochar at 2% to soil increased the crop yield by 305%, while the unmodified biochar reduced the crop yield by 60% (Kammann et al., 2015). In a pot experiment, co-composted biochar increased the total C and CEC at an application rate of 1.5%, and enhanced the crop yield by 70.8–309% as compared to the control (Luo et al., 2016). In a greenhouse experiment, the application of co-composted biochar increased the total OC by up to 212% compared to the control (Schulz et al., 2013). In a field experiment, the application of co-composted biochar at 24.2 $Mgha^{-1}$ rate significantly increased the total OC (up to 82% increase) in the topsoil as compared to that in the control or with adding only compost to the soil (Busch and Glaser, 2015).

Biochar coating with organic matter is another promising approach to enhance its efficacy in low-fertility soils. The organic materials coated on biochar surfaces act as glue for retaining dissolved nutrients in the soil (Conte and Laudicina, 2017). Hagemann et al. (2017) reported that coating the biochar surfaces with organic substances increased the mesoporosity and enhanced the potential of biochars to retain nutrients and water in the soil. However, the concept of designing suitable biochars for specific environmental issues still needs to be developed and confirmed by several field investigations.

7. Future research priorities and challenges

Biochar has been recommended as a promising soil amendment to improve soil fertility and sequester C in the soil. Several perspectives require further research to ensure the efficacy and cost-effectiveness of biochar for such purposes, particularly in the following areas:

(1) Standardization or recommendation of biochar production conditions and application rates that are more suitable for soil fertility improvement, nutrient supply to plants, and C sequestration. Those standards or guidelines will be an important help in maximizing the benefits of biochar application and in minimizing any potential environmental risks. The suggested model for biochar production standardization includes the types of feedstock, pyrolysis temperature, and pre/post-treatment of biochar. However, the relationship between feedstock and production conditions of biochar and its performance in soils still needs more documentation concerning the new advancements in biochar production methods. It remains a challenge to establish standard models for creating biochar with desired properties for specific applications in soil and the environment.

(2) Prediction of long-term decay of biochar in the field under different cropping practices. This can be achieved by investigating the decomposition rate of the stable phase of biochar in soil, which is proposed to remain in the soil for a long time (thousands of years), and setting relationships between biochar properties and its labile phase, which may quickly decompose in the soil. Any estimates of biochar stability in soil should be confirmed at the field scale; thus, long-term field experiments are very important in this aspect.

(3) Elucidation of the mechanisms of interactions between biochar, plant roots, soil organisms, and individual soil components (e.g., clay minerals, dissolved organic matter) in the

rhizosphere. This will allow us to understand the release dynamics and biogeochemical cycling of nutrients in biochar-amended soils.

(4) Determination of the adsorption-desorption capacities of biochars to soil nutrients in order to predict the nutrient bioavailability and slow release to plants in the biochar-soil complexes. However, this aspect should be tested on different biochar types applied to various soils with different properties.

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Table 1. Impact of biochar on soil fertility parameters

Feedstock	Pyrolysis temperature	Application rate	Soil type	Impact on soil properties	Reference
Wheat straw	350–550 °C	20, 40 t ha^{-1}	Anthrosol	Increased soil pH by +1.2% and +8.0% with both application rates, respectively	Zhang et al. (2010)
Sewage sludge	550 °C	50, 100 g kg^{-1} soil	Acidic soil	Both application rates increased soil pH (+20.9% and +34.1%, respectively), total carbon (+554.5% and +818.2%, respectively), and total nitrogen (+350% and +550%, respectively)	Khan et al. (2013)
Wheat straw	450 °C	10, 20, 40 t ha^{-1}	Anthrosol	Increased soil pH and soil organic carbon by +16.2, +33.2, and +51.0% with different application rates, respectively	Cui et al. (2013)
Rice straw	350–550 °C	4.5, 9 t ha^{-1}	Anthrosol	Increased organic carbon by +50% and +101% and increased total nitrogen by +9.8% and 13.4% with both application	Zhao et al. (2014)

				rates, respectively	
Crop straws	500 °C	16 t ha^{-1}	Entisol	Soil water holding capacity increased by +19.1% to +38.8%	Liu et al. (2016)
NA	400 °C	9 t ha^{-1}	Slightly acidic	Increased soil water holding capacity by +11%	Karhu et al. (2011)
Municipal biowaste	450–550 °C	40 t ha^{-1}	Anthrosol	Increased soil organic carbon by +20.2%	Bian et al. (2013)
Eucalyptus wood	350 °C, 800 °C	0, 1, 2, and 4% w/w	Ultisol	The maize biomass decreased with the biochar pyrolyzed at 800 °C (up to -25%)	Butnan et al. (2015)
Wheat straw and peanut shell	500 °C	8 t ha^{-1}	Entisol	Increased soil organic carbon (up to +56%)	El-Naggar et al. (2018b)

NA: information not available

Figure captions

Figure 1. Effects of pyrolysis temperature and biochar feedstock on biochar properties, including contents of ash (n=542), volatile matter (n=306), pH (n=358), P (n=198), C (n=615), and N (n=616). Data were obtained from the UC Davis Biochar Database, 2015.

Figure 2. Release of dissolved organic carbon (DOC), total N, and total P to solution in batch extractions of fresh biochars (a, b, and c) and aged biochars (d, e, and f) with replacement of supernatant (Reproduced from Mukherjee and Zimmerman (2013), with permission from the publisher).

Figure 3. Scanning electron micrograph images of wheat straw and wheat straw + basalt biochars produced at 550 °C. a) C-rich phase, b) accumulation and abundance of Si, Al, K, and Na, and c) abundance of Fe and O minerals inside biochar pores (Reproduced from Taherymoosavi et al. (2018), with permission from the publisher).

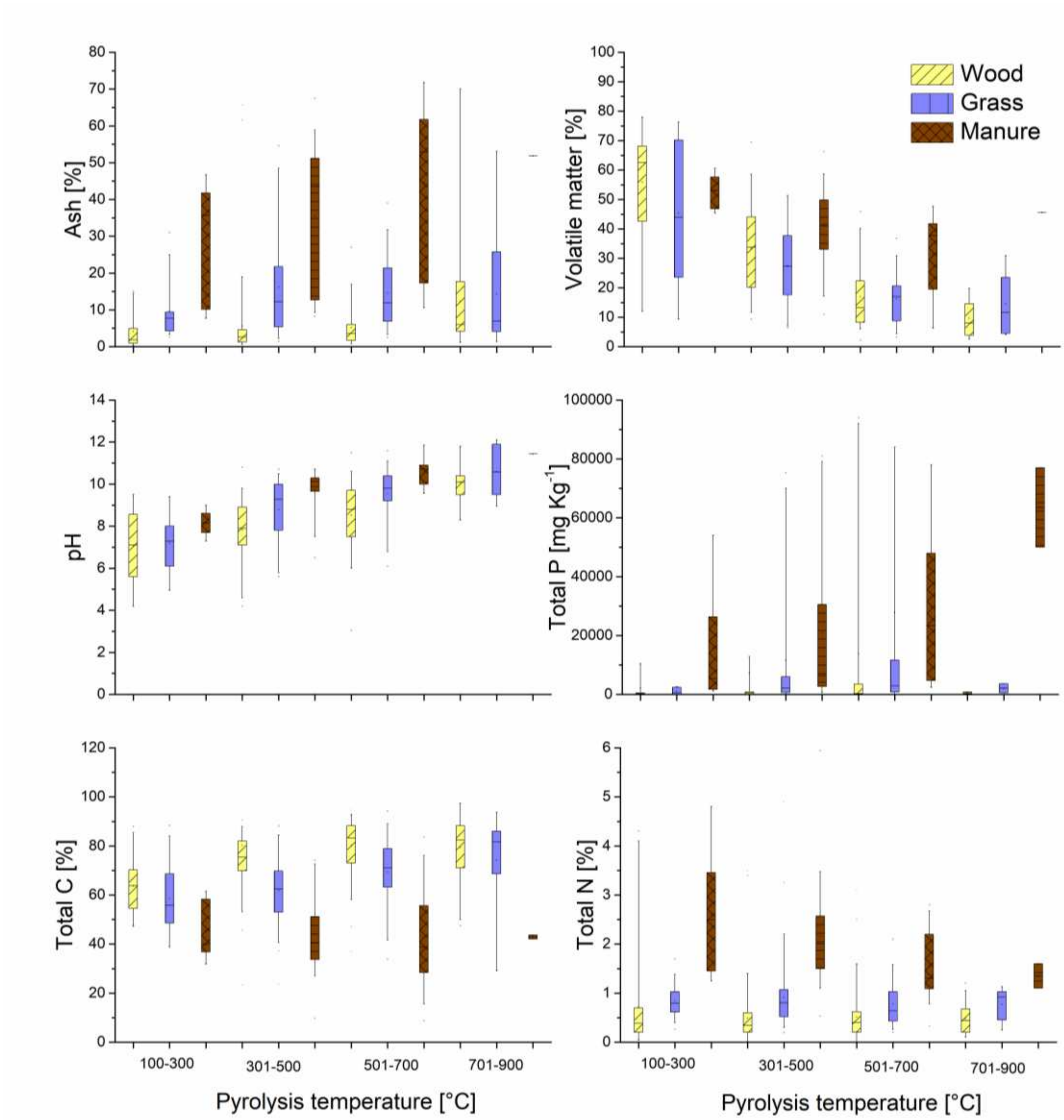
Figure 4. Scanning electron micrograph images and energy dispersive x-ray spectroscopy spectra of wheat straw + basalt biochar produced at 650 °C. Arrows represent the position of the points a and b (Reproduced from Taherymoosavi et al. (2018), with permission from the publisher).

Figure 5. Elemental mapping of wheat straw + basalt biochar produced at 650 °C for the elements a) C, b) Si, c) Al, d) Ca, e) K, f) O, g) Fe, and h) Na (Reproduced from Taherymoosavi et al. (2018), with permission from the publisher).

Figure 6. Schematic diagram of the biochar-induced priming effect on the soil. Case A shows the negative priming effect (N-PE). Case B shows the positive priming effect (P-PE).

Figure 7. Schematic diagram of biochar-induced priming effects on soils (Reproduced from Luo et al. (2017), with permission from the publisher).

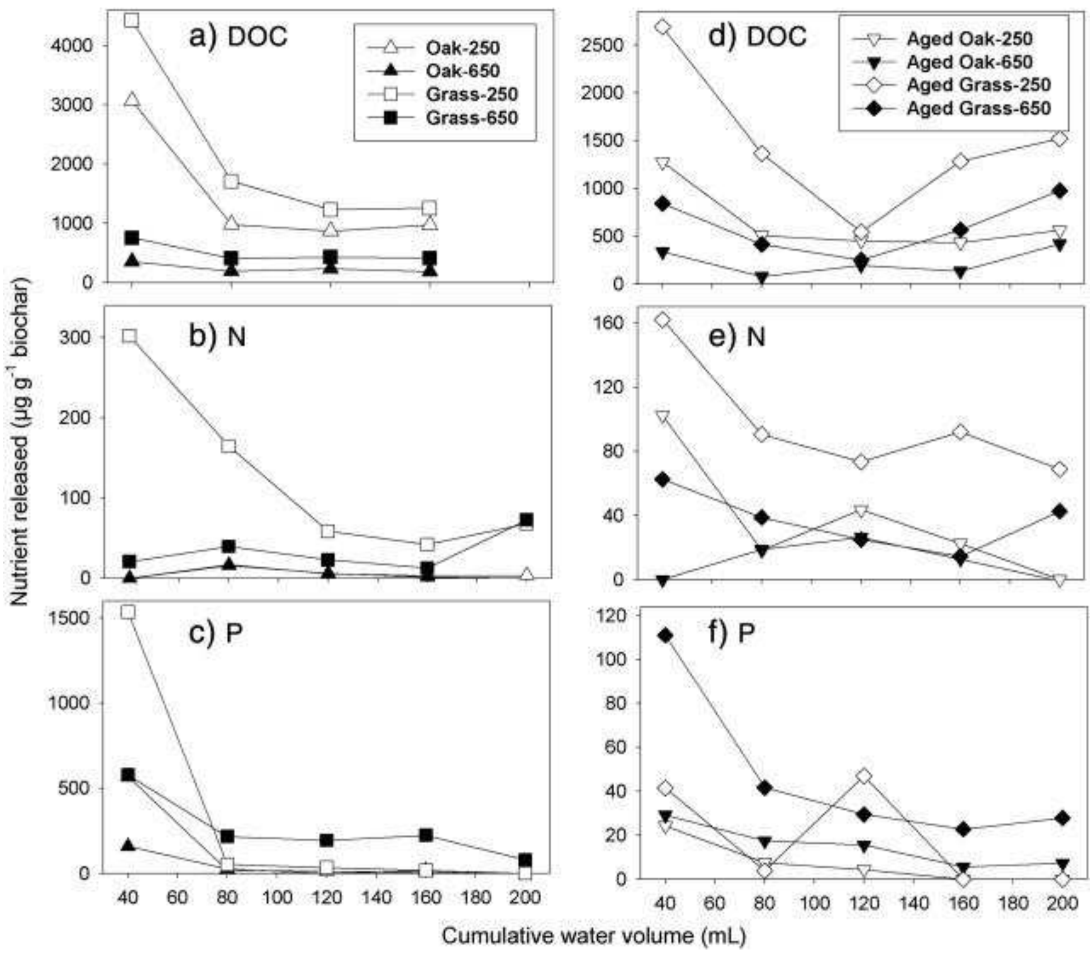
Figure 8. Cumulative CO_2-C emission from sandy and sandy loam soils treated with 30 t ha^{-1} of different biochars as compared to untreated soil (control). Error bars indicate the standard deviation of the mean. Data were obtained from El-Naggar et al. (2018d).



963

964 Figure 1.

965



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967 Figure 2.

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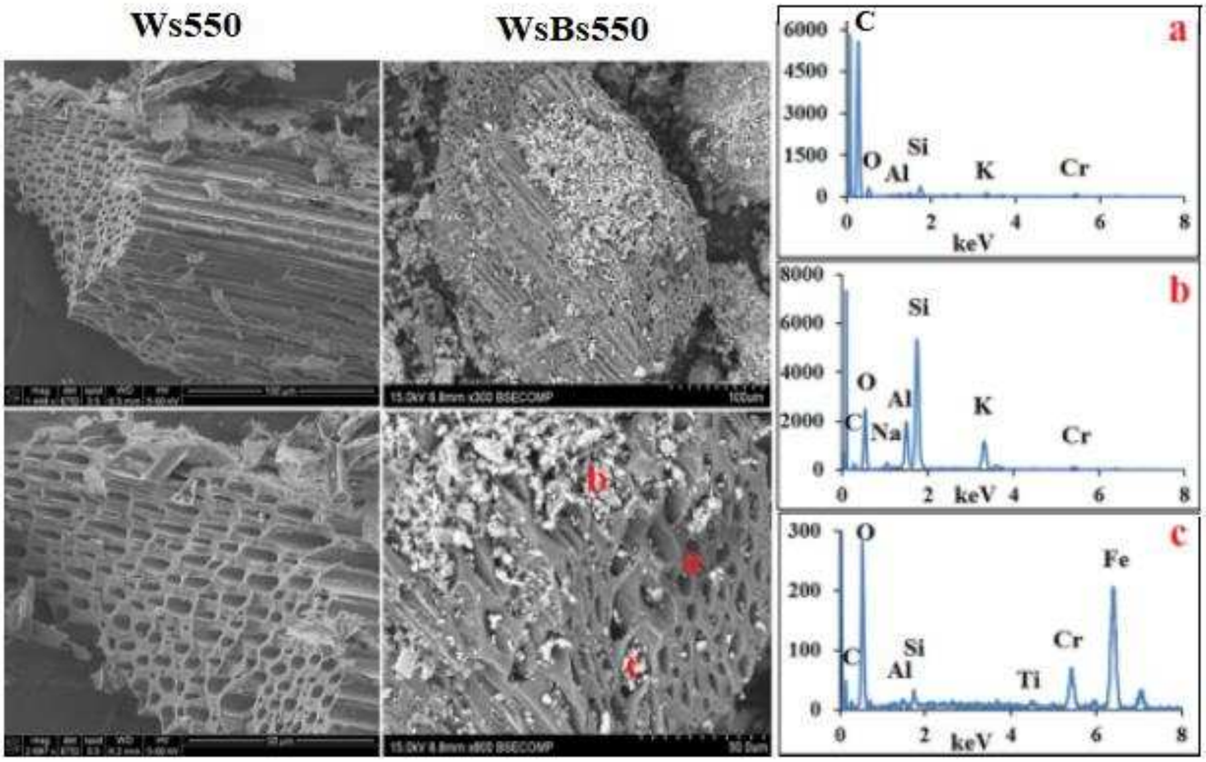

Ws550
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a
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Figure 3.

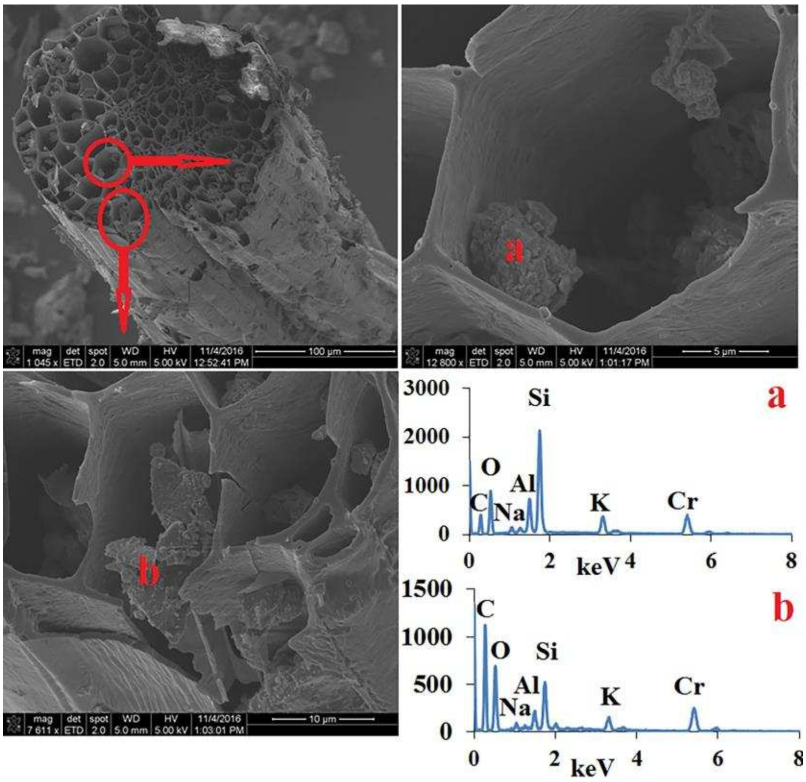
a
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Figure 4.

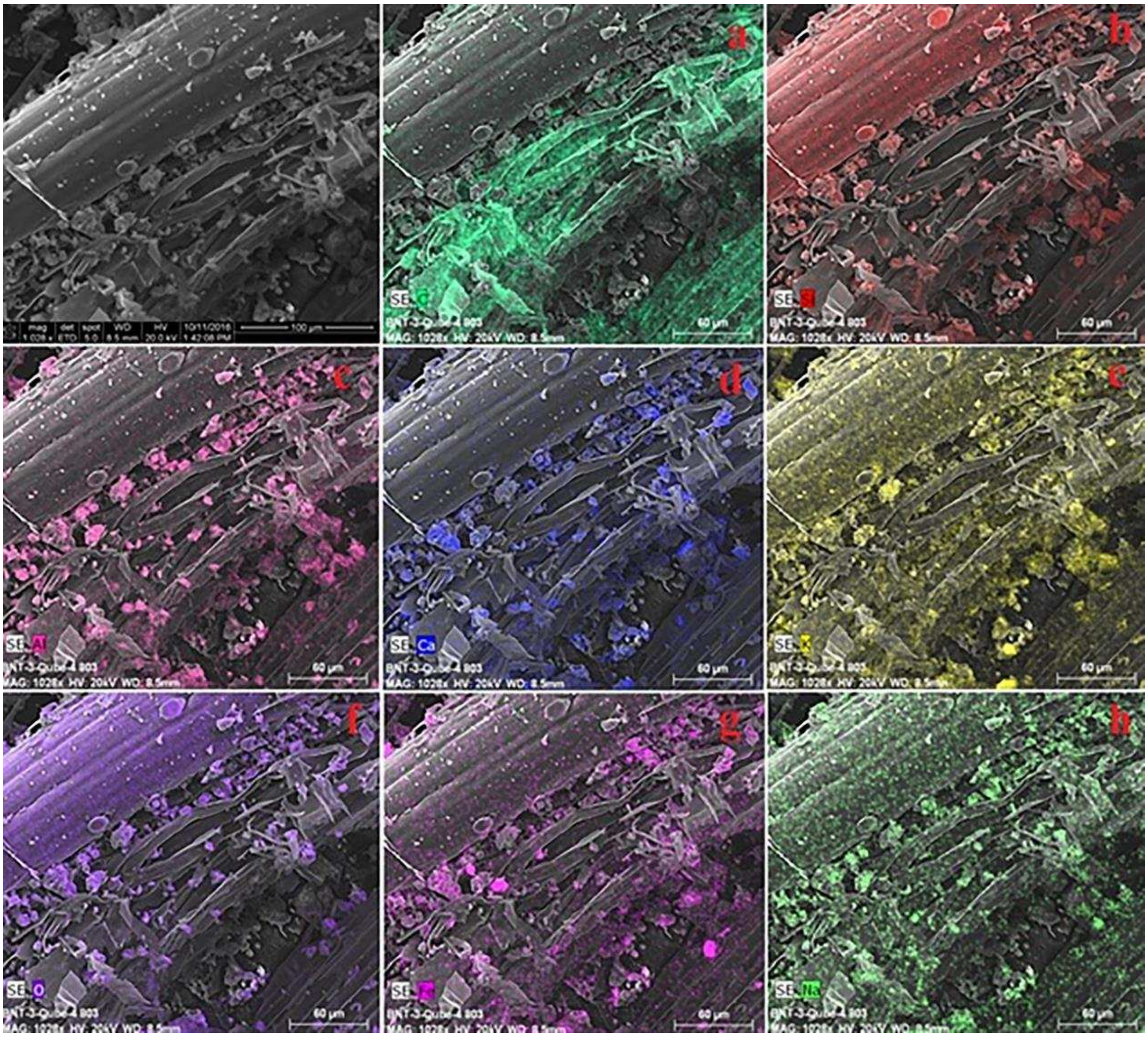

Figure 5.

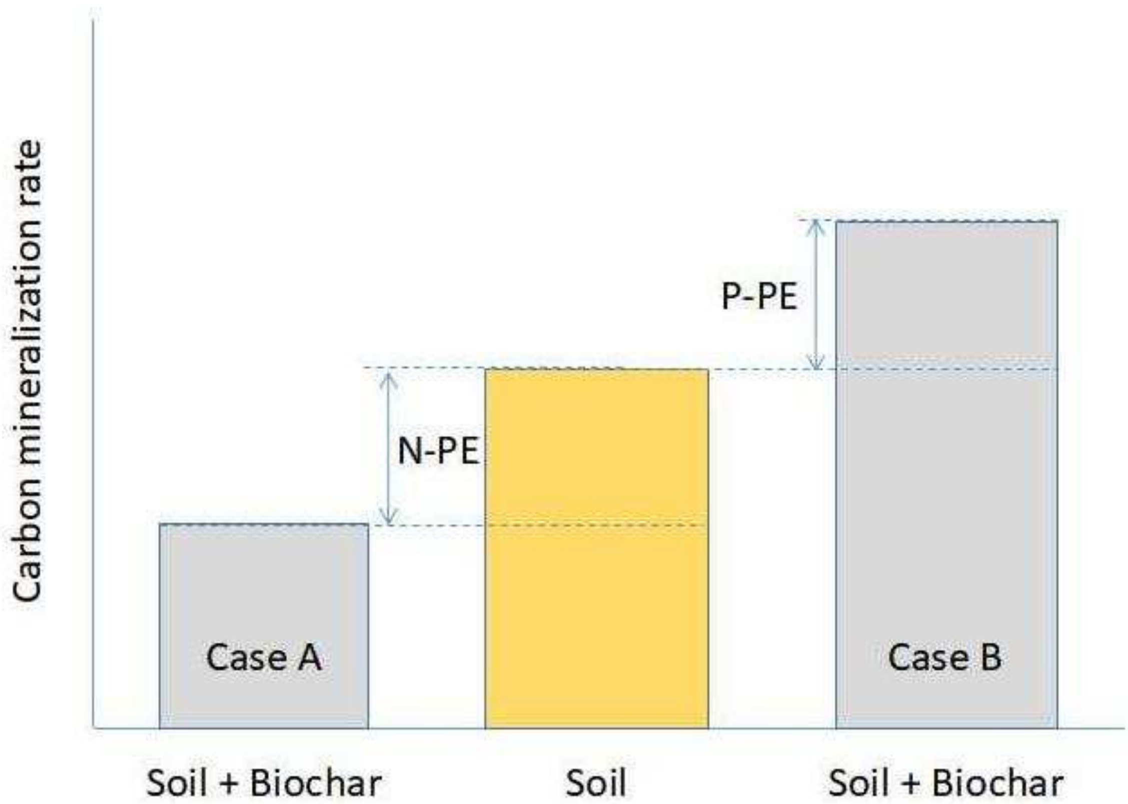
Carbon mineralization rate
P-PE
N-PE
Case A
Case B
Soil + Biochar
Soil
Soil + Biochar

Figure 6.

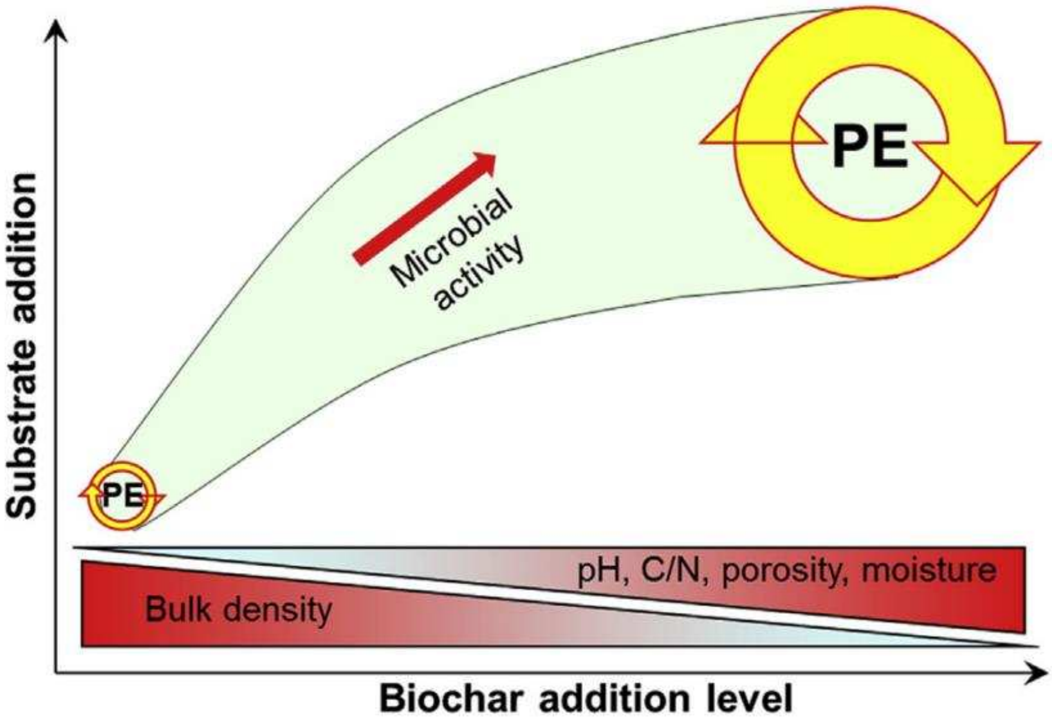
PE
Microbial activity
PE
Substrate addition
pH, C/N, porosity, moisture
Bulk density
Biochar addition level

Figure 7.

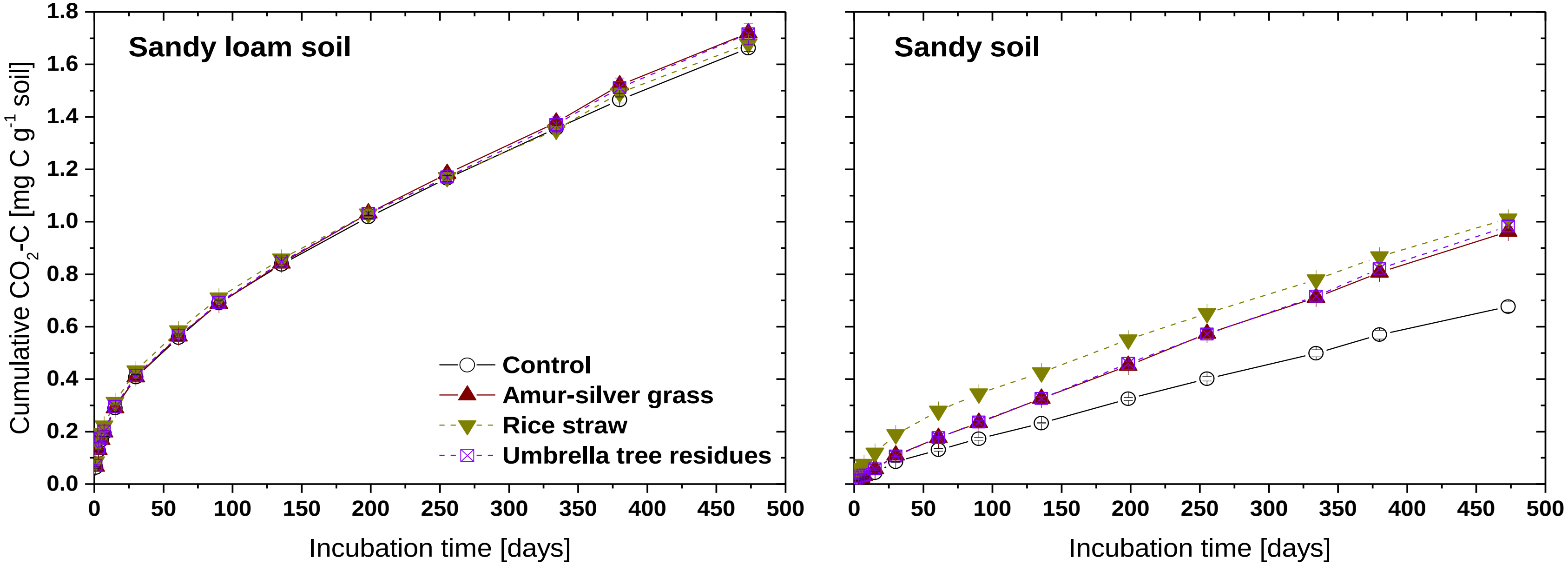
Sandy loam soil
Sandy soil
Cumulative CO2-C [mg C g-1 soil]
Incubation time [days]
Incubation time [days]
Control
Amur-silver grass
Rice straw
Umbrella tree residues
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6
1.8
0
50
100
150
200
250
300
350
400
450
500

Figure 8.