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Effects of biochar application in forest ecosystems on soil properties and greenhouse gas emissions: a review

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Abstract

Purpose Forests play a critical role in terrestrial ecosystem carbon cycling and the mitigation of global climate change. Intensive forest management and global climate change have had negative impacts on the quality of forest soils via soil acidification, reduction of soil organic carbon content, deterioration of soil biological properties, and reduction of soil biodiversity. The role of biochar in improving soil properties and the mitigation of greenhouse gas (GHG) emissions has been extensively documented in agricultural soils, while the effect of biochar application on forest soils remains poorly understood. Here, we review and summarize the available literature on the effects of biochar on soil properties and GHG emissions in forest soils.

Materials and methods This review focuses on (1) the effect of biochar application on soil physical, chemical, and microbial properties in forest ecosystems; (2) the effect of biochar application on soil GHG emissions in forest ecosystems; and (3) knowledge gaps concerning the effect of biochar application on biogeochemical and ecological processes in forest soils.

Results and discussion Biochar application to forests generally increases soil porosity, soil moisture retention, and aggregate stability while reducing soil bulk density. In addition, it typically enhances soil chemical properties including pH, organic carbon stock, cation exchange capacity, and the concentration of available phosphorous and potassium. Further, biochar application alters microbial community structure in forest soils, while the increase of soil microbial biomass is only a short-term effect of biochar application. Biochar effects on GHG emissions have been shown to be variable as reflected in significantly decreasing soil N_2O emissions, increasing soil CH_4 uptake, and complex (negative, positive, or negligible) changes of soil CO_2 emissions. Moreover, all of the aforementioned effects are biochar-, soil-, and plant-specific.

Conclusions The application of biochars to forest soils generally results in the improvement of soil physical, chemical, and microbial properties while also mitigating soil GHG emissions. Therefore, we propose that the application of biochar in forest soils has considerable advantages, and this is especially true for plantation soils with low fertility.

Keywords Biochar · Greenhouse gases · Organic carbon pool · Plantation forest · Soil acidity · Soil amendment

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1 Introduction

Mitigating increases in atmospheric CO_2 concentration is an area of growing importance and concern. More detailed understanding of how carbon cycles through various sources and sinks in different ecosystems is needed at the global scale. Forests play a key role in terrestrial ecosystem carbon cycling (Fang et al. 2001; Zhou et al. 2006b; Wood et al. 2012). The total forest area in the world decreased from 4.28 to 3.99 billion hectares from 1990 to 2015, while the area of plantation forests increased from 167.5 to 277.9 million hectares over the same time period (Payn et al. 2015). Thus, sustainable management of plantation forests is of significance for enhancing the carbon sink capacity of forest land and mitigating global climate change.

The sustainable management of plantations is affected by many climatic and environmental factors. For example, soil acidification resulting from increased nitrogen deposition plus other negative effects of global warming on soil properties and plant growth hamper the sustainable management of plantations (Bai et al. 2015; Bussotti et al. 2015). While management practices can promote the growth and productivity of forests, they can also negatively affect soil ecology (Hedwall et al. 2014). For example, management practices that were used in intensively managed Moso bamboo forests, including fertilization, tillage, and understory removal, significantly decreased soil microbial biomass C (MBC) compared to natural bamboo forests (Zhou et al. 2006a). Further, intensive management such as heavy fertilization and mulching of a Lei bamboo plantation significantly decreased the diversity of carbon sources available to microbial communities reducing native microbial diversity (Xu et al. 2008). Moreover, previous studies have demonstrated that long-term intensive management, including fertilization, tillage, and understory removal, significantly reduced the size of the total and labile organic carbon stocks in Moso bamboo and Chinese chestnut plantations (Li et al. 2013, 2014). The development of sustainable management practices, which enhance plantation productivity while impeding the degeneration of natural soils, is required.

One example of such sustainable management practices may be the application of biochar. Biochar is produced from the pyrolysis of biomass under oxygen-limited conditions (Lehmann 2007; Wu et al. 2012). Over the past decade, a considerable amount of research has demonstrated that biochar can improve soil physical properties, increase soil carbon storage capacity, stabilize soil organic carbon pools (Wang et al. 2014a), improve soil biological properties (Luo et al. 2013; Dong et al. 2015), and reduce greenhouse gas emissions (Dong et al. 2013; Gul et al. 2015; Deng et al. 2017). Thus, biochar application to forest ecosystems may be a way to restore plantation productivity in the face of global climate change and intensive management (Lehmann 2007; Stavi and Lal 2013). The majority of studies on the environmental (Wang et al. 2010; He et al. 2015, 2016a; Zhang et al. 2014, 2016a, b; Lu et al. 2014a, 2017; Yang et al. 2016, 2017; Qi et al. 2017; Yuan et al. 2017; Huang et al. 2018; Niazi et al. 2018; Xu et al. 2018) and ecological consequences of biochar application on soils have been conducted on agricultural (Wang et al. 2011; Zhang et al. 2012, 2013, 2017; Prayogo et al. 2014; Xu et al. 2015) or grassland soils (Scheer et al. 2011; Slavich et al. 2013). In contrast, the effects of biochar application on soil properties and related ecological processes in forest ecosystems have not been fully resolved. A recent review (Luo et al. 2016) summarized research on the distribution of forest wildfires and the resulting concentration and characteristics of biochar produced through these wildfires as well as its influence on soil processes. The amount of biochar produced by wildfires is estimated to be 116–385 Tg C annually (Santin et al. 2015). It results from incomplete combustion and equates to about 1% of the global aboveground plant biomass in forest ecosystems (Ohlson et al. 2009; Luo et al. 2016). Here, we review the effects of biochar applications on soil properties and processes in forest ecosystems. First, we review the effect of biochar application on the soil physical, chemical, and biological properties in forest ecosystems. Second, we discuss the potential for reducing greenhouse gas emissions via biochar application to plantations. Finally, we outline the prospects for future research concerning biochar application in forest ecosystems. The general effects of biochar application on the properties and functions in forest soils are presented in Fig. 1. In brief, when biochar is applied to forest soils, it will directly/indirectly influence soil physical and chemical properties, which will affect the soil microbial abundance, composition, and functions. The changes in the soil physical, chemical, and biological properties in turn will affect greenhouse gas emissions. The aim of this review is (i) to critically discuss the impacts of biochar application on forest soils and the mechanisms involved, and (ii) to advance the development of technologies for soil quality management and the sustainable development and management of forest soils.

2 Biochar characteristics

Biochar is produced through pyrolysis. It is carbon-rich and has a high proportion of aromatic C (Gul et al. 2015) and high chemical and biological stability (Lehmann 2007). In general, biochar has a high porosity, a large specific surface area and adsorption ability, as well as a high cation exchange capacity (Luo et al. 2016). The elemental composition of biochar generally comprises in most cases more than 60% C, N, H, and other nutrient elements including K, Ca, Na, and Mg at lower concentrations (Gul et al. 2015).

The characteristics of biochar are mainly affected by feedstock type and pyrolysis temperature (Manyà 2012; Gul et al.

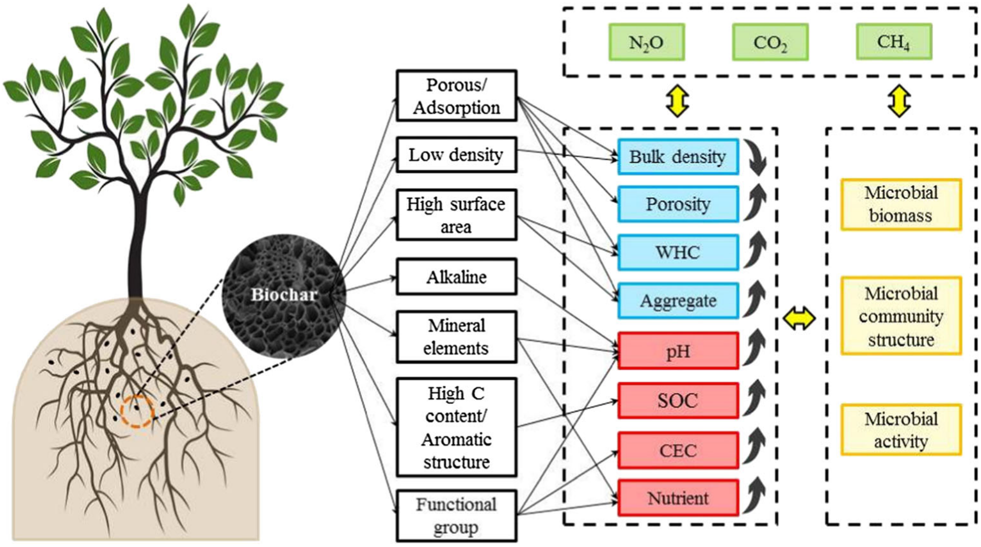

Fig. 1 Conceptual scheme for the effects of biochar application on the soil physical, chemical, and microbial properties and greenhouse gas emissions in forest ecosystems. WHC water holding capacity, SOC soil organic carbon, CEC cation exchange capacity

2015). In general, biochar derived from manures, seaweeds, and crop residues have a higher pH, a higher concentration of nutrients and less stable C compared to that derived from feedstocks with higher contents of lignocellulosics such as wood (Kloss et al. 2012; Gul et al. 2015). Additionally, biochar produced from wood biomass usually has a higher surface area than that produced from grass biomass (Mukherjee et al. 2011; Zhao et al. 2013). As to the effects of pyrolysis temperature, it has been reported that biochar produced with relatively high (600–700 °C) temperatures, leads to higher proportions of aromatic C and lower proportions of H and O functional groups, and consequently to a lower cation exchange capacity (Novak et al. 2009; Uchimiya et al. 2011; Ahmad et al. 2014). On the other hand, biochar produced with relatively low (300–400 °C) pyrolysis temperature has leads to a higher proportion of C-O and C-H functional groups and consequently a higher cation exchange capacity (Glaser et al. 2002; Novak et al. 2009).

3 Effects of biochar on soil physical properties

Soil physical properties, mainly including bulk density, soil structure, water holding capacity and aggregate formation and stability, directly/indirectly influence the retention, movement and availability of soil nutrients as well as soil microbial activity, and consequently affect plant growth (Alameda et al. 2012; Cardoso et al. 2013; Nawaz et al. 2013; Hartmann et al. 2014; Kormanek et al. 2015; Heydari et al. 2017). Forest clear cutting, converting natural forests into plantations, and the intensive management of plantations have been shown to deteriorate soil physical health, i.e., increasing bulk density, reducing porosity, and deteriorating soil structure (Jorge et al. 2012; Kleibl et al. 2014; Sankura et al. 2014; Chang et al. 2017; Tonks et al. 2017). Finding a cost-effective way to counteract these undesirable effects of forest plantation management on soils is critical. Due to the typical characteristics of biochar including its high porosity, low density, and high surface area, biochar application generally decreases soil bulk density, while increasing soil porosity, water holding capacity, and aggregate stability (Fig. 1). In the following section, we discuss the effects of biochar application on soil physical properties in forest ecosystems and the mechanisms involved.

3.1 Soil bulk density

Biochar can effectively reduce soil bulk density, and this has been demonstrated in a wide variety of agricultural soils of differing textures (Artiola et al. 2012; Abel et al. 2013; Basso et al. 2013; Herath et al. 2013; Castellini et al. 2015; Burrell et al. 2016). In comparison, reports on biochar effects on the bulk density of soils under forests are scarce. A few case studies showed effects comparable to those observed in agricultural systems (Prober et al. 2014; Mertens et al. 2017). For example, Prober et al. (2014) reported that soil bulk density was significantly lower on plots treated with a greenwaste biochar at a rate of 20 t ha^{-1} in mesic woodlands after 2 years compared to the control. The authors attributed the observed decrease to the incorporation of the low bulk density biochar material into the soil (Prober et al. 2014).

However, in other studies, biochar application had no significant effect on bulk density. Mertens et al. (2017) showed that *Prosopis juliflora*-biochar application did not significantly affect the soil bulk density of a sandy soil planted with seedlings of *Spondias tuberosa* Arruda. The authors hypothesized that this lack of effect was due to the low number of pyrogenic nanopores of the biochar, which could be explained by the low pyrolysis temperature used in their experiment. In conclusion, it should be noted that differences in biochar materials, pyrolysis processes, application rates, and soil types present in different forest systems will affect changes in soil bulk density. The large difference between root growth of forest plants and agricultural crops necessitates further study regarding the influence of biochar on soil bulk density in forest ecosystems.

3.2 Soil porosity

Numerous studies have shown that the application of biochar can improve soil porosity and structure, and consequently increase the water holding capacity of soils and improve nutrient bioavailability (Moyano et al. 2013; Gul et al. 2015), which promotes plant growth (Xiao et al. 2016a; Obia et al. 2016). Soil pores can be classified into the following five categories according to their equivalent pore diameter (EPD): macropores (75–100 μm), mesopores (30–75 μm), micropores (5–30 μm), ultramicropores (0.1–5 μm), and crytopores (0.1–0.007 μm) (Cameron and Buchan 2006). The pore size distribution affects aeration, water holding capacity, and drainage capacity of soils.

Recent studies have demonstrated that the effects of biochar application vary with differences in soil pore characteristics (Hseu et al. 2014; Lu et al. 2014b; Sun and Lu 2014). For example, Hseu et al. (2014) reported that the abundance of macropores, defined in this study as pores with a diameter > 75 μm, increased by 4 to 27% in mudstone slopeland soils after rice husk biochar application (2.5, 5, and 10%, *w*/*w*) and incubation for 168 days at 28 °C and 60% water holding capacity. The authors reasoned that this result could be attributed to soil particle rearrangement. They also found that micropores, defined as pores with diameters between 5 and 30 μm, increased from 11 to 54%. Similarly, Lu et al. (2014b) found that the application of rice husk biochar (2, 4, and 6%, *w*/*w*) significantly increased the porosity of clayey soils. The authors hypothesized that the increase in macropore volume (> 75 μm) was due to the biochar's high abundance of macropores. Further, addition of straw biochar, woodchips biochar, and wastewater-sludge biochar increased the total soil porosity in a clayey soil (Vertisol) by 29, 12, and 16%, respectively (Sun and Lu 2014). The results from this study confirm that the pore structure of biochar greatly influenced the effect of biochar on soil pore structure (Sun and Lu 2014).

Despite the abundance of biochar studies on soils, little analyses have been conducted on forest soil porosity. Pore size distribution in forest soils is influenced by soil biological properties, vegetation, and forest management methods. Therefore, it is critical to quantitatively assess the pore size distribution of forest soils, for example, by using X-ray computed tomography in order to predict the structural transformation of these soils upon biochar addition. Studies to this effect will help to better understand the changes in soil processes induced by biochar application and provide an important basis for future soil improvement and enhanced forestry production.

3.3 Soil water holding capacity

Forest soils act as moisture reservoirs and regulators in the forest water cycle. Numerous studies have shown that soil water stress affects the growth of natural and planted forests and increases tree mortality (Brzostek et al. 2014; Fargeon et al. 2016). Recent studies have demonstrated that the application of biochar can significantly increase soil water holding capacity and thus moisture contents in forest ecosystems. For example, Prober et al. (2014) reported that the soil moisture content in mesic woodlands increased by 6 to 25% after the application of greenwaste biochar at a rate of 20 t ha^{-1}. Similarly, Li et al. (2017b) reported that the overall average runoff decreased by 28% after the application of rice straw biochar at a rate of 20 t ha^{-1} over a period of 2 years, compared to a control treatment. The reduction in runoff was attributed, inter alia, to the strong water retention effect of biochar.

In addition, the response of soil hydrological properties to biochar applications is biochar-specific (Lewis et al. 2006; Mukherjee and Lal 2013). For example, Uzoma et al. (2011) investigated the effects of biochar application at various rates (0, 10, and 20 t ha^{-1}) and biochar pyrolysis temperatures (300, 400, and 500 °C) on the hydraulic properties of sandy soils. The experiments indicated that biochar application increased the saturated water content from 0.2 to 56.1%. The application of biochar significantly increased the available water capacity of all treated sandy soils compared to the control except for the treatment with biochar prepared with a pyrolysis temperature of 300 °C and an application rate of 10 t ha^{-1}. In addition to biochar specificity, the response of soil hydrological properties to biochar amendments is soil-specific (Lewis et al. 2006). For example, biochar treatment significantly increased the water holding capacity of sandy soils, but no such effect was observed for silt loams despite equivalent water potentials (Tian et al. 2015). In conclusion, the water holding capacity of forest soils plays an important role for forestry production, especially in arid and semi-arid areas. It is critical to further study the effects of different biochar types on the water holding capacity and plant available soil water contents of different soil types and under different forest ecosystems, as well as the associated mechanisms.

3.4 Soil aggregation

In general, the application of biochar has been shown to positively affect soil aggregate characteristics (Ding et al. 2016). For example, Lei and Zhang (2013) reported that in an incubation experiment with a loamy soil biochar addition (5%, *w*/w) significantly promoted the formation of macro-aggregates and enhanced the stabilization of macro-aggregates in the first

30 days. Lu et al. (2014b) similarly found that the addition of rice husk biochar (2, 4, and 6%, *w*/*w*) increased soil aggregation by 8–36% in a clayey soil (Vertisol). Likewise, Ouyang et al. (2013) found that the addition of dairy manure biochar (2%, *w*/*w*) significantly promoted the formation of macroaggregates in both a silty clay soil and a sandy loam soil. In addition to the above studies, field and incubation studies have demonstrated enhanced soil aggregation through biochars, produced at pyrolysis temperatures between 400 and 600 °C, in sandy loam to clay loam textured soils (Ibrahim et al. 2013; Mukherjee and Lal 2013; Khademalrasoul et al. 2014; Soinne et al. 2014).

The mechanisms underlying the increased aggregate formation and stability due to biochar can be explained by several processes:

1. The physical and chemical nature of biochars affects soil aggregation formation. For example, the first step in the process of soil aggregate formation and stabilization, the binding of biochar to organo-mineral complexes, is affected by the surface area of the biochar and its O/C ratio (Gul et al. 2015). This mechanism is supported by the observation that biochar produced at high temperature (700 °C) with a low O/C ratio did not change the aggregation characteristics of a coarse-textured soil (Busscher et al. 2010, 2011).
2. The carboxyl groups formed by the surface oxidation of biochar can complex with soil minerals, and thereby enhance the stability of soil aggregates (Glaser et al. 2002).
3. Biochar application increases root biomass and root activity, thus favoring fungal growth, which will enhance the stability of soil aggregates (Bossuyt et al. 2001; Bruun et al. 2014).
4. Lastly, biochar treatment increases soil hydrophobicity, and consequently, decreases the extent of clay swelling and aggregate disruption, thereby improving the stability of soil aggregates (Lu et al. 2014b).

In contrast to the above studies, others have shown that biochar application had no significant effect on aggregate stability. For example, Busscher et al. (2010) reported that the addition of walnut shell biochar at a rate of 0, 5, 10, and 20 g kg^{-1} (incubated at 10% soil moisture content at temperatures ranging from 17 to 27 °C) had no significant effect on the stability of the aggregates in a loamy sand. Likewise, Peng et al. (2011) reported that the addition of 1% straw biomass carbon (incubated in a darkened room of 25 °C, 40% field capacity) had no significant effect on the stability of soil aggregates in Ultisol soils in southern China. The authors attributed the no effect of biochar application on the soil aggregation to the fact that the biochar did not produce aggregating exudates during its mineralization (Peng et al. 2011). Thus, the biochar processing parameters, soil type, and climate conditions can all significantly influence the effect of biochar on soil aggregates (Burrell et al. 2016).

4 Effects of biochar on soil chemical properties

The growth and production of forests are closely associated with soil chemical properties, such as pH, cation exchange capacity (CEC), organic C pool, and nutrient status. Recently, long-term intensive plantation management, mainly including chemical fertilization and understory removal, has been reported to decrease soil pH and deplete plantations' soil organic C pools (Li et al. 2013, 2014), which negatively affect the growth of forest plants (Lapenis et al. 2004; Ito et al. 2011; Lorenz and Lal 2014). Biochar addition can directly influence soil chemical properties (Fig. 1), since it is an alkaline material containing various mineral elements and a large proportion of carbon with highly aromatic structures, as well as a large number of functional groups, such as COO^-, on its surface (Lehmann et al. 2011; Luo et al. 2016). In addition, biochar application will affect soil nutrient availability and transformation indirectly through altering soil physical properties (Gul et al. 2015). In the following sections, we discuss the effects of biochar application on different soil chemical properties and the mechanisms involved.

4.1 Soil pH

Increased soil pH as a result of biochar application has been extensively investigated in agricultural soils (Jeffery et al. 2011; Gul et al. 2015), and similar results have been found in forest soils (Wang et al. 2014b; Wrobel-Tobiszewska et al. 2016). For example, Wrobel-Tobiszewska et al. (2016) found that high rates of biochar application (50–100 t ha^{-1}) increased soil pH from 4.0 to 4.8 in a *Eucalyptus* forestry plantation. Further, Rhoades et al. (2017) reported that the joint application of biochar (application rate of 20 t ha^{-1}) and mulch (application rate of 37 t ha^{-1}) increased soil pH from 5.7 to 6.4 in a lodgepole pine (*Pinus contorta*) forest, while the application of either biochar or mulch alone had no pH effect. Our previous results (Wang et al. 2014b) showed that the application of bamboo leaf biochar (application rate of 5 t ha^{-1}) significantly increased soil pH in an intensively managed Chinese chestnut plantation grown on a Ferrasol. There are two possible mechanistic explanations for the observed increases in soil pH as a result of biochar application. First, biochar is alkaline and contains mineral carbonates with an abundance of basic-charged groups (Yuan and Xu 2011). Thus, the observed increase in soil pH may be simply due to the addition of alkaline material. Alternatively, biochar application decreases the exchangeable aluminum content of soils through binding Al^{3+} ion by oxygenated functional groups on its surface, thereby increasing the abundance of soil exchangeable base cations, increasing soil base saturation, and ultimately resulting in a soil pH increase (Yuan and Xu 2011; Yuan et al. 2011; Dai et al. 2017). However, these mechanisms require further investigation as some studies have shown no pH effect of biochar application to forest soils (Noyce et al. 2015; Sackett et al. 2015; Mitchell et al. 2016). These contrasting findings among different studies may be attributed to differences in biochar feedstock, the pyrolysis process, and distinct soil properties in addition to those of the local environment (Dai et al. 2017).

Biochar has recently gained attention as an excellent alternative to lime amendment, which is the most commonly used method of increasing soil pH (Dai et al. 2017). Biochar has the additional advantage over lime amendment of enhancing soil carbon sequestration. In contrast, lime decomposes after incorporation into soils, and subsequently is one of the primary sources of soil CO_2 emissions (West and McBride 2005). Biochar, on the other hand, can play a role in mitigating soil greenhouse gas emissions (discussed in the following section). The utility of various biochars for increasing soil pH necessitates research into the optimization of biochar production for lime-like effects.

4.2 Soil organic carbon pools

Increasing soil organic carbon levels by incorporating biochar into soils may be an effective way to mitigate soil organic carbon depletion in intensively managed ecosystems. Laird et al. (2010) showed that soil organic carbon content in a Clarion soil (Mesic Typic Hapludolls) increased with the addition of biochar after adding 0, 5, 10, and 20 g kg^{-1} biochar to soils. Further, our investigation (Wang et al. 2014b) showed that biochar application at a rate of 5 t ha^{-1} significantly increased soil organic carbon storage in a Chinese chestnut plantation, but addition of bamboo leaf with an equivalent amount of organic carbon did not have a comparable effect. Potentially, the primary reason for these observations is that the carbon present in biochar is stable and difficult to decompose in soil environments, thus contributing to the recalcitrant soil carbon pool (Lorenz and Lal 2014). Hamer et al. (2004) reported that the decomposition rates of biochar (10%, *w*/*w*) made from corn straw, rye straw, and wood were 0.78, 0.72, and 0.26%, respectively, after incubation for 60 days (20 °C, 60% of water holding capacity). Moreover, Kuzyakov et al. (2009) used ^{14}C-labeled ryegrass to produce biochar, and found that only 1.8–2.1% of the biochar was decomposed after 60 days of incubation. These results point to the long-term stability of biochar-amended carbon in soil environments. However, it is necessary to investigate longer-term effects of biochar applications on the soil carbon pool and subsequent transformations in future studies.

Biochar application also impacts the labile fractions of the organic carbon pools in soils (Luo et al. 2011; Lorenz and Lal 2014). Wang et al. (2014b) reported that biochar application significantly increased the concentrations of soil water-soluble organic carbon and MBC in a Chinese chestnut plantation in the first month after application. Such short-term effects can probably be attributed to the release of oil condensates that are formed during pyrolysis (Smith et al. 2010). However, the effects of biochar application on soil labile organic carbon significantly differ depending on soil types. For example, Durenkamp et al. (2010) found that the application of biochar increased soil MBC content in clay soils, while it decreased soil MBC content in sandy soils. Moreover, others have found that the effect of biochar application on soil labile organic carbon was dependent on the time elapsed since the biochar application. For example, Hua et al. (2012) reported that coconut shell biochar application initially increased the soil labile organic carbon, but this effect gradually decreased with time.

4.3 Soil cation exchange capacity

Biochar application generally increases soil CEC, which improves plant nutrient availability and is thus beneficial for plant growth (Atkinson et al. 2010; Lorenz and Lal 2014). Glisczynski et al. (2016) found that the application of 30 t ha^{-1} biochar-compost substrates significantly increased soil CEC in poplar, willow, and alder tree plantations grown on a Luvic Stagnosol (Episiltic). Further, Cheng et al. (2008) reported that the incubation (30 °C and 60% of water holding capacity) of oak biochar over 1 year increased soil CEC from 1.7 to 71.0 mmol kg^{-1}, which is likely because of the continuous oxidation of functional group on the surface of the biochar surface and the adsorption of organic acids by the biochar. The authors also suggested that soil CEC will further increase over time in the biochar-amended treatments. In contrast, Novak et al. (2009) demonstrated that the application of walnut shell biochar (pyrolysis temperature of 700 °C) had no significant effect on the CEC of acidic sandy soils. Reasons might have included the short experimental time (120 days) and the low oxidation extent of the biochar used.

The increase in soil CEC resulting from biochar applications can be explained via two possible mechanisms. First, biochar adsorbs soil organic matter and other compounds, and this capacity increases with the degree of biochar oxidation. Adsorption to biochar increases charge density, and consequently enhances soil CEC (Liang et al. 2006; Lee et al. 2010; Van Zwieten et al. 2010a). Second, biochar gradually oxidizes after its application to soil, and as a consequence, aromatic rings are replaced by COO^{-} functional groups, and the overall surface negative charge increases on the biochar, thereby enhancing soil CEC (Mao et al. 2012). Fresh biochar generally has far fewer COO^{-} constituents and a lower CEC compared to oxidized biochar, which supports the above argumentation (Brewer et al. 2009; Nguyen et al. 2010).

The chemical composition and structure of biochar can significantly affect its capacity to improve CEC (Van Zwieten et al. 2010a; Mao et al. 2012). Lee et al. (2010) found that the biochar O/C ratio is related to soil CEC capacities. Higher O/C values indicate a higher hydroxyl group content, where carboxylates and carbonyl groups of biochars improve soil CEC. The physical and chemical properties of biochars are closely related to pyrolysis temperatures (Angın et al. 2013). For example, higher pyrolysis temperatures (e.g., 600 °C) resulted in a greater surface area and consequently reduced charge density, leading to a lower CEC, in addition to fewer volatile compounds (Lehmann et al. 2011), which also contributed to the overall negatively charged components. Further, Wang et al. (2017) proposed that increasing pyrolysis temperatures leads to a loss of functional groups, which thereby decreases biochar CEC. In addition to the above, the effect of biochar application on soil CEC is dependent on soil type. For example, biochar application resulted in larger CEC increases in acidic soils than in calcareous soils (Liang et al. 2006; Gul et al. 2015).

4.4 Soil nutrient availability

Biochar application is thought to increase the inorganic nutrient content and bioavailability since biochar itself also contains various inorganic constituents (Biederman and Harpole 2013). Biochar produced from wood waste materials generally contains high levels of soluble potassium and variable concentrations of phosphorus and calcium (Page-Dumroese et al. 2015). Sackett et al. (2015) showed that bioavailable potassium concentrations significantly increased in the initial period (2–6 weeks) after maple biochar application at a rate of 5 t ha^{-1} in a northern hardwood forest soil, while the concentrations of available calcium and magnesium increased 9 to 12 months following application. In addition, Gundale et al. (2016) reported that biochar application at a rate of 10 t ha^{-1} in a boreal forest increased the soil's net N mineralization rate and NH_4^+ concentration after two growing seasons. Other studies have shown that biochar application increased other nutrient concentrations including silica, boron, and molybdenum (Kloss et al. 2014; Liu et al. 2014). Kloss et al. (2014) found that biochar application (3%) in a greenhouse pot experiment significantly increased boron and molybdenum availability for three different soil types (Planosol, Cambisol, and Chernozem).

The amount and type of nutrients present in biochar is related to the feedstock type (Gaskin et al. 2008) in addition to pyrolysis temperature and duration (Tsai et al. 2007), indicating that different biochar types will affect nutrient enhancement in different ways. Moreover, increases in soil nutrients caused by biochar application are generally short-lived, and such effects decline with time due to plant uptake and leaching (Topoliantz et al. 2005; Steiner et al. 2007; Gaskin et al. 2010). Lehmann et al. (2003) proposed that the immediate beneficial effects of charcoal application on plant growth and yield in tropical soils can be attributed to the increased bioavailability of Ca, Cu, K, P, and Zn. However, the effect of biochar application on plant growth and yield over the long term is mainly due to modifying nutrient bioavailability rather than via the direct supply of nutrients from biochar (Glaser et al. 2002). For example, biochar application increased phosphorous and potassium retention through adsorption to its large and porous surface, and consequently decreased nutrient leaching losses. This process increased the concentrations of available phosphorous and available potassium in the soil, which then increased plant growth and yield (Biederman and Harpole 2013). In summary, a combination of biochars and chemical fertilizers is needed to ensure sustainable forest management in the long term, as nutrients in biochar alone are not sufficient to maintain the growth of plantation trees.

5 Effects of biochar on soil microbial properties

The above discussed changes in soil physical and chemical properties caused by biochar application will also change soil microbial properties (Gul et al. 2015). In general, the positive changes such as decreased bulk density and increased porosity, water holding capacity, pH, and nutrient pools, will also have positive effects on soil microbial abundance and activity (Lehmann et al. 2011; Gul et al. 2015). In addition, because of the specific characteristics of biochar including its high surface area, high porosity and abundance of pores of various sizes, biochar will provide a habitat for soil microorganisms, and will promote their growth (Fig. 1). The following sections are dedicated to the effects of biochar on soil microbial biomass and community structure.

5.1 Microbial biomass

The application of biochar has long been assumed to significantly influence soil MBC level. However, a consensus has not been reached regarding the effects of biochar on soil MBC (Gul et al. 2015). Biochar addition at various rates (0.45 and 2.27%, *w/w*) to a coarse textured soil decreased MBC after 10 weeks of a pot experiment (soil incubated between 13 and 25 °C, and at 50% of the soil's water holding capacity) (Dempster et al. 2012). But in long-term field experiments, it was found that soil MBC concentrations either increased or did not change significantly after several years of regular biochar applications (Jones et al. 2012; Rousk et al. 2013; Zheng et al. 2016). A previous meta-analysis including different soil and land use types conducted by Biederman and Harpole

(2013) suggested that biochar application resulted in a significant increase in soil MBC. Recently, another meta-analysis across soils under different land uses suggested that MBC content increased on average by 18% (12–23%) after biochar application across 395 paired observations (Liu et al. 2016).

However, changes in MBC were quite variable depending on biochar type and application rate, land use type, experimental method, soil texture, and fertilization management (Liu et al. 2016). Among these factors, application rate could be one of the most important drivers for the different responses of MBC to biochar application. Unlike in agricultural systems, where biochar application rates of biochar are usually larger than 10 t ha^{-1} and applications are frequent in order to increase plant productivity and crop yields, application rates in forest systems are generally much lower. Although the application of biochar to forest ecosystems has received less attention compared with agricultural systems, a few studies suggest that biochar amendment at a low rate did not change microbial biomass in forest soils (Noyce et al. 2015; Wang et al. 2014b). For example, Noyce et al. (2015) found that biochar application at a rate of 5 t ha^{-1} had no effect on soil MBC concentration in a tolerant hardwood forest after 1 or 2 years of biochar applications. Another study by Wang et al. (2014b) also showed that biochar application at a rate of 5 t ha^{-1} increased soil MBC concentration only in the first 2 months after application in a Chinese chestnut plantation soil. Domene et al. (2014) found that low application rates (1 and 3 t ha^{-1}) of biochar did not result in changes of soil MBC concentrations, while the high application rate of 30 t ha^{-1} significantly increased soil MBC concentration. Similarly, Maestrini et al. (2014) found that the application of ryegrass-derived biochar increased microbial biomass at an application rate equivalent to 27 t ha^{-1} after a 158-day incubation period time (incubation at 27 °C and 70% of water holding capacity) in a Cambisol forest soil. All of the abovementioned results indicate that the effect of biochar application on soil MBC concentration are generally short-lived and rate-dependent.

The short-term effects of biochar application on enhancing MBC can be attributed to the release or dissolution of biological oil condensates formed during pyrolysis and present in biochar particles (Smith et al. 2010). These release and/or dissolution processes frequently occur in the first 2 months after biochar application. In addition, it is also possible that the porous structure of biochar material may potentially provide a suitable habitat for microorganism, such as providing labile available organic C and increasing water retention, acting as a refuge, and protecting microorganisms from predators (Lehmann et al. 2011; Luo et al. 2013; Quilliam et al. 2013; Chen et al. 2013, 2015). However, other studies argued that biochar amendment at relatively high rates may have a negative effect on soil MBC, as larger amounts of biochar with high C/N ratio tended to induce the immobilization of soil microbial N, reducing microbial activities (Ameloot et al. 2013). Nevertheless, more research is needed to investigate the reasons for such contradictory results and the mechanisms of how biochar affects MBC.

5.2 Microbial community structure

Soil microbial community structure is also assumed to be significantly affected by the application of biochar (Gul et al. 2015; Luo et al. 2017a). However, the effects of biochar on fungal and bacterial abundance and diversity patterns do not follow clear trends. Mitchell et al. (2015) reported that biochar application at 10 and 20 t ha^{-1} resulted in a significant increase of the bacterial/fungal ratio and a decreased Gram-negative/Gram-positive bacteria ratio in a temperate forest soil. In a 6-month laboratory incubation study, Santos et al. (2012) observed that biochar was utilized by all bacteria groups, especially Gram-positive bacteria, in temperate forest soils. Based on DNA sequencing technique, Khodadad et al. (2011) also found that application of oak and grass biochar significantly increased the abundance of several actinobacterial families (i.e., Gram-positive bacteria) in a Floridian forest soil. It is not surprising that Actinobacteria were stimulated by biochar applications as members in this group play a role in pyrogenic C metabolism and grow readily on carbon-rich refractory materials (O’Neill et al. 2009). These results indicate that biochar application may favor the growth of bacteria, in particular Gram-positive bacteria. Similar results have also been observed by Chen et al. (2013), who found that biochar amendments increased bacterial but decreased fungal gene abundance in a rice paddy soil.

The varied responses of bacterial and fungal communities to biochar applications might be closely related to their ecological characteristics and functioning, because bacteria and fungi differ strongly in their nutrient demand, turnover rate, and stress tolerance, for example, resilience to pH and water stress (Rousk et al. 2009). Compared with fungi, bacteria are more sensitive to labile substrate (Khodadad et al. 2011; Lehmann et al. 2011). Therefore, the labile C in biochar could be an important driver for bacterial growth directly after biochar applications (Ameloot et al. 2013; Farrell et al. 2013). Besides, the typical size of fungal hyphae is generally larger than bacteria, which may prohibit them from colonizing micropores. As a result, bacteria could be better protected from grazing than fungi, especially in smaller pores (Chen et al. 2013). Increase in soil pH may also play a key role in regulating microbial abundance and diversity, since neutral or slightly alkaline conditions are known to favor bacterial growth (Rousk et al. 2009).

A few researchers stated that biochar could also enhance fungal growth and activity, because fungi have the ability to colonize on carbon materials with low quality, such as char with a high proportion of aromatic C compound (Hockaday et al. 2007; Jin 2010; Lehmann et al. 2011; Li et al. 2017a).

Supporting these statements, Mitchell et al. (2016) showed that fungal PLFA concentrations increased significantly 3 years after biochar was applied together with P fertilizer in a P-limited temperate hardwood forest in Ontario, Canada. Steinbeiss et al. (2009) showed that the application of biochar that was pyrolyzed from yeast promoted fungi and significantly decreased the bacterial/fungal ratio in both agricultural and forest soils, while glucose-derived biochar stimulated Gram-negative bacteria, suggesting that the responses of bacteria and fungi to biochar amendment vary and could depend on biochar feedstock. Moreover, our study (Chen et al. 2017) found that bamboo biochar increased fungal PLFA concentrations and the fungal/bacterial ratio, changed microbial community structure in a bamboo plantation soil, but such effects were largely dependent on the application rate and biochar particle size. Some wood-decaying fungal species can utilize biochar as a carbon substrate, thus enhancing their growth (Fontaine et al. 2011). The increased fungal/bacterial ratio may imply a change in microbial function towards decreased carbon loss because a fungal-dominated microbial community is believed to improve carbon use efficiency (Lehmann et al. 2011; Chen et al. 2013). Intriguingly, using the same study system as Mitchell et al. (2015), Noyce et al. (2015) found that biochar application at a rate of 5 t ha^{-1} had no significant effect on the bacterial and fungal community compositions or the fungal/bacterial ratio. The difference in results might be due to the application of biochar to the soil surface (0–20 cm) by Noyce et al. (2015), while the biochar was thoroughly mixed into the soil in the incubation study of Mitchell et al. (2015). These results highlight that the effect of biochar application on soil microbial community structure is complicated, especially since biochar applications significantly influence soil physical and chemical properties, which may then lead to complex interactions affecting soil microbial community characteristics (Gul et al. 2015).

6 Effects of biochar on soil greenhouse gas emissions

Soil carbon dioxide (CO_2) emissions, also known as soil respiration, form the main pathway for soil organic carbon to enter the atmosphere. It is the primary mechanism of carbon loss from terrestrial ecosystems contributing to climate change (Peng et al. 2008; Xu and Shang 2016). Furthermore, methane (CH_4) and nitrous oxide (N_2O) are regarded as major greenhouse gases, and their global warming potential per unit mass are 25 and 198 times that of CO_2, respectively, at the century scale (IPCC 2014). As outlined above, the incorporation of biochar into soil can change soil physical, chemical, and biological properties significantly, and consequently, it has also significant effects on the soil's greenhouse emissions (Fig. 1). In the following sections, we consider how biochar application affects soil CO_2, N_2O and CH_4 emissions, and discuss the mechanisms involved in the response of soil greenhouse gas emissions to biochar application.

6.1 Soil CO_2 emission

Biochar addition significantly affects soil water content, porosity, pH, cation exchange capacity, carbon and nitrogen dynamics, and plant productivity, which all can have a significant effect on soil CO_2 emissions (Jones et al. 2011; Stavi and Lal 2013; Luo et al. 2017b). However, the effect of biochar application on soil CO_2 fluxes in forest ecosystems varies considerably among studies. The application of biochar to forest soils increased, decreased, or had negligible effects on CO_2 emissions (Table 1). Mitchell et al. (2015) reported that the application of sugar maple biochar (5, 10, and 20 t ha^{-1}) significantly increased soil CO_2 emissions in a temperate forest soil. Johnson et al. (2017) also reported increased CO_2 emissions from a biochar-amended soil compared to the emissions from the untreated control soil. This was corroborated by the findings of Hawthorne et al. (2017), who reported significantly higher CO_2 fluxes from a Douglas-fir forest soil treated with 10% biochar compared to the emissions from the same soil treated with 1% biochar. The lowest CO_2 flux was observed from the untreated control soil. In contrast, Sun et al. (2014) reported that the application of biochar (30 t ha^{-1}) significantly reduced CO_2 emissions by 31.5% from pine forest soils. Malghani et al. (2013) found in an incubation experiment that biochar application did not affect the CO_2 emissions from a forest soil. Similarly, field studies conducted by Wang et al. (2014b), Sackett et al. (2015), and Zhou et al. (2017) also reported that biochar applications did not influence soil CO_2 fluxes from forest soils. The variable effects of biochar application on soil CO_2 flux in the above studies might be explained by differences in the type and rate of biochar, vegetation, and soil types, in addition to the time period between CO_2 measurement and biochar application (Sohi et al. 2010; Ennis et al. 2012; Woolf and Lehmann 2012).

The mechanisms underlying the effects of biochar application on soil CO_2 emissions can be generally encapsulated by four processes: (1) the biochar itself contains labile organic carbon which contributes to the labile organic carbon pool in soils after its application, thereby promoting soil CO_2 emissions (Yoo and Kang 2012; Mukherjee and Lal 2013; Spokas 2013); (2) biochar has a large adsorption capacity that can affect soil surface CO_2 emissions by adsorbing soil CO_2 molecules (Kasozi et al. 2010; Liang et al. 2010); (3) biochar application to soil influences soil physical and chemical properties including water content, porosity, aggregation, pH, and CEC (Liang et al. 2010; Jones et al. 2011), which then indirectly affect CO_2 emissions; and (4) biochar application can significantly affect the diversity and activities of microbial taxa that are involved in CO_2 production(Liu et al. 2009; Liu

Table 1 Effects of biochar application on the greenhouse gas emissions in forest soils

Soil type	Study type (scale)	Biochar type	Biochar rate	Time	CO_2 emission (over control)	CH_4 uptake (over control)	N_2O emission (over control)	Reference
Cambisols	Laboratory	Corn silage (500 °C)	1%, *w*/*w*	105 days	No significant difference	No significant difference	Decreased N_2O emission	Malghani et al. (2013)
Ferralsols	Laboratory	Chicken manure (540 °C)	10%, *w*/*w*	84 days	–	Increased CH_4 uptake	–	Yu et al. (2013)
Brunisol	Laboratory	Sugar maple wood (500 °C)	5, 10, and 20 t ha^{-1}	24 weeks	Increased CO_2 emission	–	–	Mitchell et al. (2015)
Humo-ferric podzols	Laboratory	Douglas-fir (420 °C)	1 and 10%, *w*/*w*	25 days	Increased CO_2 emission	Decreased CH_4 uptake	No significant difference in the 1% biochar treatment; Increased N_2O emission by 191% in the 10% biochar treatment	Hawthorne et al. (2017)
Lixisol	Field	Wheat straw (450 °C)	30 t ha^{-1}	1 year	Decreased CO_2 emission by 31.5%	–	Decreased N_2O emission by 25.5%	Sun et al. (2014)
Ferralsols	Field	Bamboo leaf (500 °C)	5 t ha^{-1}	1 year	No significant difference	–	–	Wang et al. (2014)
Humo-ferric podzols	Field	Mixed maple and spruce sawdust (350–450 °C)	5 t ha^{-1}	1 year	No significant difference	No significant difference	No significant difference	Sackett et al. (2015)
Ferralsols	Field	Bamboo leaf (500 °C)	5 t ha^{-1}	1 year	–	–	Decreased N_2O emission by 20.5%	Xiao et al. (2016b)
Humo-ferric podzols	Laboratory	Douglas-fir slash (420 °C)	20 t ha^{-1}	3 months	Increased CO_2 emission by 6.6%	Decreased CH_4 uptake by 8.4%	–	Johnson et al. (2017)
Ultisol	Field	Chicken manure (400 °C)/Sawdust (400 °C)	24 t ha^{-1}	1 year	–	No significant difference	No significant difference	Lin et al. (2017)
Ferralsols	Field	Bamboo (800 °C)	10 and 30 t ha^{-1}	16 months	No significant difference	–	–	Zhou et al. (2017)

et al. 2011; Zhang et al. 2012; Mitchell et al. 2015; Wang et al. 2016).

6.2 Soil CH_4 emission

Biochar has been shown to reduce CH_4 emissions from waterlogged rice paddies (Liu et al. 2011) and to enhance CH_4 uptake in aerobic soils (Karhu et al. 2011). The few published studies on the application of biochar to forest soils suggest that biochar reduces soil CH_4 emissions. Yu et al. (2013) reported that the application of chicken manure biochar (10%, *w*/*w*) significantly increased CH_4 uptake in forest soils. In a field study, Xiao (2016) also showed that, regardless of the application rate, biochar treatment significantly increased soil CH_4 uptake in an intensively managed Chinese chestnut plantation.

There are two primary mechanisms underlying this observed increased soil CH_4 uptake. First, biochar application generally increases soil pH which favors the growth of methanotrophy (Inubushi et al. 2005; Jeffery et al. 2016). Second, biochar application decreases soil bulk density and increases soil porosity, which favors CH_4 oxidation and uptake activity by soil bacteria (Brassard et al. 2016). Increased soil aeration and porosity induced by biochar application and the consequent promotion of oxic conditions generally decrease CH_4 production, because CH_4 oxidation is an aerobic metabolic process, that is dependent on oxygen availability (Brassard et al. 2016).

Most studies have confirmed a reduction in CH_4 emissions from biochar-amended soils, but a few studies have suggested that the application of biochar did not affect soil CH_4 emissions (Malghani et al. 2013; Sackett et al. 2015; Lin et al. 2017), and in some cases, even reduced soil CH_4 uptake (Hawthorne et al. 2017). Malghani et al. (2013) found that the addition of corn silage biochar (1%, *w*/*w*) had no significant effect on the CH_4 emissions from a deciduous forest soil. A recent study also reported that there was no significant difference in CH_4 flux between biochar-treated and control soils in a temperate hardwood forest (Sackett et al. 2015). Similarly, Lin et al. (2017) reported that either sawdust biochar application (24 t ha^{-1}) or chicken manure biochar application (24 t ha^{-1}) did not affect CH_4 uptake in a subtropic acidic forest soil. In contrast, Hawthorne et al. (2017) showed that biochar application (1 or 10% additions, *w*/*w*) significantly decreased soil CH_4 uptake, with net CH_4 oxidation decreasing concomitantly with increasing biochar application rates. Therefore, the mechanisms underlying CH_4 flux from soils after biochar application, and especially concerning microbial metabolism, require further investigation.

6.3 Soil N_2O emission

Evidence concerning the potential for reduced N_2O emissions as a result of biochar application in dryland soils has been confirmed in many farmland soils (He et al. 2016a, b). Though limited in number, investigations of N_2O emissions after biochar application in forest environments have indicated significantly reduced N_2O emissions (Malghani et al. 2013; Sun et al. 2014; Xiao et al. 2016b). Sun et al. (2014) found that the application of biochar (30 t ha^{-1}) to a pine forest soil significantly decreased (25.5%) cumulative N_2O emissions. Malghani et al. (2013) also reported that the application of corn silage biochar (1%, *w*/*w*) to deciduous forest soils significantly decreased the N_2O emissions in a spruce forest soil. Likewise, Xiao et al. (2016b) showed that biochar application at a rate of 5 t ha^{-1} to a Chinese chestnut forest reduced annual average flux and annual cumulative total soil N_2O emissions by 27.4 and 20.5% for, respectively, compared to the control.

There are two predominant mechanisms for these reductions. First, biochar application significantly enhances soil aeration and oxygen concentration in the soil profile, which in turn inhibits soil denitrification by microorganisms, which primarily occurs under sub-oxic conditions (Yanai et al. 2007; Van Zwieten et al. 2010b). Second, biochar application results in the adsorption of NH_4^+ and/or NO_3^- to biochar particles, which then enhances plant growth, reduces NH_3 volatilization, or immobilizes nitrogen compounds. These processes decrease the inorganic nitrogen pool available for nitrifiers or denitrifiers, which produce N_2O as a metabolic byproduct (Singh et al. 2010; Van Zwieten et al. 2010b; Clough et al. 2013).

In contrast to the above results, positive or non-significant effects have also been reported in some studies. Hawthorne et al. (2017) found that the application of 10% biochar in a forest soil significantly increased N_2O emissions, while no significant effect was observed after the application of 1% biochar. Sackett et al. (2015) found that the application of biochar (5 t ha^{-1}) in a temperate hardwood forest did not change soil N_2O emission. Thus, the effects of biochar addition on soil N_2O emission processes are quite complicated, and the mechanisms involved require further investigation.

7 Prospects for biochar application to forest soils

The application of biochar in forest soils has a number of benefits, including increased soil pH, CEC content, aggregate stability, and organic carbon stock, while reducing soil bulk density and soil greenhouse gas emissions. Such effects are biochar-, soil-, and plant-specific and can vary considerably among systems. In addition, the application of biochar is also regarded as an important practice for remediating soils contaminated with heavy metals and organic compounds (Zhang et al. 2013). Therefore, the application of biochar in forest soils can have considerable benefits. This is especially true for plantation soils with low organic carbon contents and slight to moderate levels of contamination (Thomas and Gale 2015). However, some issues still remain unsolved and the mechanisms by which biochar application affect soil processes are yet unclear and require further investigation. Below, we outline five specific recommendations to guide future investigations about the effects of biochar applications to forest soils.

1. Recent studies have demonstrated that biochar application can significantly affect microbial community structure and diversity in forest soils. However, little information is available regarding the effect of biochar application on specific functions that are conducted by soil microorganisms and their gene functions that are related to carbon cycling (e.g., *cbbL*, *cbhI*, *cel*5, and *lcc*) and nitrogen cycling (e.g., *nir*K, *nir*S, *nos*Z, and *nar*G), when compared to those of agricultural soils. Assessing the effects of biochar addition on the aforementioned microbial functions will help us better understand the mechanisms by which biochar affects the biogeochemical processing of soil nutrients in forest ecosystems.
2. The impact of biochar addition on soil respiration components (i.e., autotrophic respiration and heterotrophic respiration) is still poorly understood, although overall respiration rates in forest ecosystems are well-studied (Wang et al. 2014b; Mitchell et al. 2015). Estimating the net primary productivity (NPP) and net ecosystem productivity (NEP) of forest ecosystems requires quantification of autotrophic respiration and heterotrophic respiration (Gower et al. 2001). Therefore, it is necessary to separately quantify soil respiration components during the assessment of biochar application effects on forest ecosystem soil carbon cycling.
3. Biochar can have considerable advantages for improving soil physical, chemical, and biological properties. However, biochar application alone is not sufficient to meet the nutrient needs for tree growth and productivity. Thus, the study of new fertilizers (e.g., those rich in biochar and a mixture of certain nutrients) may be an effective way to mitigate depletion of soil organic carbon pools and the large nutrient requirements that are typical of intensively managed plantations.
4. The application of biochar can directly affect tree growth, but it can also indirectly affect tree growth by modifying soil properties. Significant genotypic responses may differ among tree species related to types and rates of biochar applications. Thus, it is critical to conduct studies on the effects of different types of biochar on the growth of different tree species.
5. Most of the studies regarding the effects of biochar application on soil properties were conducted through short-term incubation experiments that lasted mostly less than 3 years (Nguyen et al. 2017). The carbon in biochar is primarily composed of aromatic carbon molecules with soil residence times that can exceed 10 or even 100 years. Therefore, it is also critical to examine the long-term effects of biochar addition on soil properties and greenhouse gas emissions, which is of great significance for the effective application of biochar in forest soils.
6. Assessing costs and benefits of biochar applications is complicated. Considering only the nutrient value of biochar, the costs of biochar applications exceed those of fertilizer applications. But as outlined in this review, biochar applications have many additional benefits including the improvement of soil properties and the mitigation of greenhouse gas emissions. In contrast, long-term chemical fertilization has negative effects on soil properties and carbon sequestration. However, as far as we know, no cost-benefit analysis for biochar application in forest ecosystems has been conducted yet. Prior to promoting field-scale applications of biochar in forest ecosystems, a comprehensive cost-benefit analysis of biochar application in forest ecosystems is required.
7. Although most case studies have reported positive effects of biochar on soil properties in forest ecosystems, there might be some so far less studied adverse effects of biochar on forest soils. Recently, it has been reported that during the production of biochar, other materials, such as polycyclic aromatic hydrocarbons (PAHs) and volatile organic compounds (VOCs), were produced and remained on the surface of biochar particles (Dutta et al. 2017). Such types of materials would negatively affect the soil microbial community and the growth of tree plants (Dutta et al. 2017). Therefore, the ecotoxicological effects of biochar application on the growth of soil microorganisms and tree plants in forest ecosystems need to be investigated in future study.

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